8-2011

Ecologically-Based Manipulation Practices for Managing Bromus tectorum-infested Rangelands

Beth Fowers
Utah State University

ECOLOGICALLY-BASED MANIPULATION PRACTICES FOR MANAGING BROMUS TECTORUM-INFESTED RANGELANDS

by

Beth Fowers

A thesis submitted in partial fulfillment
of the requirements for the degree

of

MASTER OF SCIENCE

in

Range Science

Approved:

Dr. Christopher A. Call Major Professor	Dr. Thomas A. Monaco Committee Member
Dr. Corey Ransom Committee Member	Dr. Mark R. McLellan Vice President for Research and Dean of the School of Graduate Studies

UTAH STATE UNIVERSITY
Logan, Utah

2011

ABSTRACT

Ecologically-Based Manipulation Practices for Managing

Bromus tectorum-infested Rangelands

by

Beth Fowers, Master of Science

Utah State University, 2011

Major Professor: Christopher A. Call
Department: Wildland Resources

Cheatgrass (*Bromus tectorum*) is an invasive annual grass common in several semiarid plant communities in the western U.S. *B. tectorum* presence increases fire frequency and size, reducing species diversity, and leading to annual species-dominated systems with inconsistent livestock forage potential and degraded wildlife habitat value. Most efforts to manage *B. tectorum*-dominated rangelands have focused on controlling the plant itself rather than addressing the causes of vegetation change. An alternative approach, ecologically-based invasive plant management (EBIPM), identifies treatments that can alter factors associated with the causes of succession, leading to a more desirable vegetation state. This study utilized the EBIPM framework to design a large-scale demonstration project, which implemented a series of manipulation treatments (mowing, prescribed fire, imazapic herbicide, and seeding with perennial species) to suppress *B. tectorum* and promote desirable species. The treatments were implemented at two semiarid shrubland sites in northwestern Utah. Treatments were evaluated by measuring resident vegetation cover, density, aboveground biomass, and litter and soil seed banks. Herbicide was most effective in reducing *B. tectorum* cover, density, and biomass, while fire was effective in reducing seed density in the litter seed bank. Treatment interactions were rarely significant; however, by combining fire and herbicide, increased *B. tectorum* control was achieved. Seedlings of seeded perennial grasses emerged in all treatments; however, establishment by the end of the first growing season was greatest in treatments involving fire. The results of this study indicate that using a decision-making framework to select a series of treatments that alter the causes of succession can improve the management of *B. tectroum*-dominated rangelands.

(116 pages)

ACKNOWLEDGMENTS

I would like to thank all those who helped me with this complicated project; I could not have finished without help. I would like to especially thank my major advisor, Chris Call, who helped with all the planning and many of the details involved with this constantly adapting project as well as providing much needed editing for this final paper. I would also need to thank my committee members for the support and help they offered as well: Tom Monaco and Corey Ransom. I received so much help from the USDA research station especially from Tom's technician Justin Williams who offered support, and advice, and was part of the implementation of the treatments and was present for some of the data collection. This project would not have occurred without his help. I also have to include Merilynn Hirsch with my thanks as well because even though she also had her data collection occurring at the Park Valley sites, I feel that we were able to support and help each other no matter if it was in the field or back on campus working through data and processes. It was definitely a strong experience working with her. I was also grateful for the support and advice offered by a fellow graduate student of Dr. Call's, Jamin Johanson, both ecologic and educational. Then I can't forget the students who supported the project through their work: the Range Club putting up our fences, Rob Watson and Casey Snider for their help the first year of data collection, and Kyle Becker the second year, as well as Trent Staheli and Kevin McArthur for their help with greenhouse and sample processing. Finally I want to mention my family for the support and love they constantly offered, thank you.

Beth Fowers

CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

Bromus tectorum, commonly known as cheatgrass, downy brome, or Junegrass, is an invasive annual grass that has spread over 23 million hectares across the West (Stewart and Hull 1949; Rice 2005). It was introduced to the United States in the mid 1800's from Eurasia, likely as a contaminant of crop seed and alfalfa, and continued to spread through agricultural practices and grazing (Young and Allen 1997). By the 1920's it had spread to most of its current range and had invaded semiarid rangelands wherever perennial cover was disturbed (Stewart and Hull 1949). *B. tectorum* was instrumental in altering the community structure of many areas and continues today to be a management concern, as well as a somewhat-valued forage species.

Historically, much of the Great Basin contained big sagebrush (*Artemisia tridentata*)/bunchgrass communities composed of shrubs (primarily non-sprouting shrubs), perennial grasses and forbs, and some annual species, none of which were well adapted to season-long grazing by large herbivores (Young et al. 1972). With the introduction of domesticated livestock in the Great Basin, perennial understory species were weakened by excessive grazing, and *B. tectorum*, which had already evolved to fill niches left open through domesticated grazing, was able to invade (Morrow and Stahlman 1984; Young and Allen 1997). The advantage that was gained through historical grazing practices has been continued or exacerbated through rest-rotation grazing systems which allow *B. tectorum* the chance to grow and set seed (Young and Allen 1997). *B. tectorum* was also dispersed by migratory bands of grazing sheep, and advanced from railroad right-of-ways because of fires caused by engine brakes (Young et al. 1972; Young and Allen 1997). Fires historically occurred every 50 to 70 years in sagebrush/grasslands in the Great Basin, and with the invasion and expansion of *B. tectorum* the fire frequency in these community types has decreased to 2 to 5 years (Young et al. 1972; Young and Evans 1978; Whisenant 1990). The effects of *B. tectorum* invasion into native perennial communities can also be exacerbated by seed-eating rodents and birds, which have preferences for seeds of perennial species and not annual grasses (Rice 2005).

With the spread of *B. tectorum*, previously grazed species became less common and livestock managers began to utilize the annual species for forage. Even though *B. tectorum* is an important forage species, it is unreliable because of the large fluctuations in production based on yearly precipitation, such as the 10-fold yield difference in consecutive years found by Hull and Pechanec (1947), as well as having the disadvantage of a short green forage period (Stewart and Hull 1949; Morrow and Stahlman 1984). While *B. tectorum* can be used for forage, other species, especially perennial grasses, are more desirable because of their longer green forage periods and the reduced risk of frequent, large fires (Stewart and Hull 1949; Young et al. 1987; Young and Allen 1997). Additionally, the sharp awns on *B. tectorum* spikelets can cause damage and sores to grazing animals' eyes, mouths, nostrils, and intestines (Leopold 1949: USDA-NRCS 2009).

Ecological and Economic Impacts

B. tectorum plants are more flammable than other species because they dry out sooner and produce a continuous fuel to carry fire (Stewart and Hull 1949). In some areas, fires have spread from dense stands of *B. tectorum* to nearby intact sagebrush communities (Stewart and Hull 1949). Fires can increase in number and size, causing ecological damage and increasing fire suppression and rehabilitation costs (Stewart and Hull 1949; Knapp 1996). Fires cause all species to restart their growth, some through resprouting and others from seed. While annual species are adapted to this, perennial species take longer to recover from disturbance (Stewart and Hull 1949; Young and Evans 1973). Fires in *B. tectorum* systems typically occur after seed set of annuals, but before seeds have matured in many perennial plants, which reduces the ability of perennial species to survive in that area because recovery is based solely on adult plants resprouting or viable seeds remaining in the seed bank (Young et al. 1972). These stresses, in conjunction with unmanaged grazing, can negatively impact perennial plants and eventually eliminate them from the system.

The successional pathway in disturbed sagebrush communities typically moves from bare soil after a fire to annual forbs such as mustards (*Sisymbrium altissimum*) and Russian thistle (*Salsola iberica*). Later, other early successional species such as *B. tectorum* become common as residual perennial species are out-competed and the early annual forbs are pushed out (Stewart and Hull 1949; Young et al. 1972). Additionally, *B. tectorum* invaded systems have the possibility of becoming invaded by other non-native species that are of more concern, such as medusahead (*Taeniatherum caput-medusae*) and Scotch thistle (*Onopordum acanthium*) (Young and Evans 1978; Rice 2005).

One of the concerns with *B. tectorum*-infested rangelands is the possibility of erosion, especially with the amount of bare soil exposed after fires. With heavy grazing, plants allocate more resources to regrowing leaf and reproductive material than root growth, and less biomass carries over and accumulates as litter (Facelli and Pickett 1991; Hild et al. 2001). As fire enters a system, there is also a reduction in the litter layer which helped protect the soil surface. However, with managed grazing and in the absence of fire, *B. tectorum* can also form enough root mass and produce litter which work to hold soils in place in what otherwise would be a degraded system (Stewart and Hull 1949; Morrow and Stahlman 1984).

The altering of plant community structure affects the diversity of native and domesticated animal species. Rather than being able to utilize native plant species with higher nutritional content through the fall and winter, species such as elk and deer must rely on dead *B. tectorum*, which loses much of its nutritional value when it dries (Stewart and Hull 1949). As diverse perennial communities become converted to annual grasslands, other native species, such as sage grouse and Brewer's sparrow lose their habitats. The prey base of raptors can also be threatened because of the decreased ability of *B. tectorum*-dominated areas to support rodent populations (Rice 2005).

Major economic impacts are associated with *B. tectorum* invasion. In agricultural systems many crops such as alfalfa and winter wheat exhibit reduced yields when *B. tectorum* is present, costing millions annually (Rice 2005). On rangelands, perennial species which would have been used as forage throughout the year are replaced by an annual grass which can only be used early in the year before it dies. However, the cost of most concern on rangelands occurs because of the increase in fire frequency and size with the presence of *B. tectorum*. This cost is associated with fire suppression and rehabilitation, as both require large financial and resource inputs (Rice 2005). In 2005, the Interior Agencies combined spent $29.2 million on all levels of fire management related training (Quadrennial Fire and Fuel Review Report 2005). Chambers (2008) stated that over $19 million was being spent annually on restoring sagebrush ecosystems.

Seed mixes alone can also be very expensive, such as a stabilization seed mix composed of 8 species costing $162/ha, or a fire prevention mix composed of 6 species costing $128/ha, each with only 2 native species, which keeps the costs at a minimum (Gary Kidd, Utah BLM, personal communication February 21, 2008). Epanchin-Niell et al. (2009) created a model to predict the costs and long-term benefits of possible restoration plans for both *B. tectorum*-dominated landscapes and for native plants. This model showed that a combination of treatments is still best and managers need to be wary of the continually changing conditions associated with *B. tectorum* abundance. Additional costs are associated with damage to human life, property, or other commodities, especially when fires approach the wildland/urban interface (Zouhar et al. 2008).

Characteristics of *B. tectorum*

The biology, life cycle, and growth requirements of *B. tectorum* have been studied in depth (Thill et al. 1984). *B. tectorum*, as a winter annual, germinates from fall to spring when precipitation and temperature conditions are conducive to growth. Plants that germinate in the fall or early winter grow until cold temperatures retard growth; the plants then overwinter as seedlings until temperatures rise. Plant roots continue to grow, even through the cold winter months, allowing them an advantage over other species without the ability to withstand the cold soil temperatures (Harris 1967). The continual growth of the roots allows plants to have an extensive root system by the time aboveground plant growth can start utilizing water and nutrient resources early in the spring, and reducing surface soil moisture rapidly (Harris 1967; Melgoza et al. 1990). This limits the water available for perennial species later in the season when *B. tectorum* has reached maturity and is drying. The roots of *B. tectorum* individuals are typically 33 cm deep, but have been found to extend 150 cm deep; while some plants formed a dense sod with their root growth (Hulbert 1955; Harris 1967; Young and Evans 1973; Thill et al. 1984).

B. tectorum begins the boot stage, when spikelets begin to form, in May and begins to senesce and dry by June. Reproduction is completely by seed, and the plants are largely self-pollinated. Under specific environmental conditions, typically when resources are released or disturbance reduces *B. tectorum* density, hybridization can occur and produce genotypes adapted to different microsites (Young and Evans 1978; Evans and Young 1984). *B. tectorum* seed shatter occurs earlier in the season, and the flammability period begins 4 to 6 weeks earlier than for perennial species, which are still susceptible to seed loss from fire before maturing (Stewart and Hull 1949). Spikelets and individual florets (hereafter referred to as seeds) are dispersed at the time of seed shatter. Florets are 0.9 – 1.5 cm long, with an awn 0.5 – 1.8 cm long, and spikelets are 1.2 – 2.0 cm long, excluding awns (Stewart and Hull 1949; Stubbendieck et al. 2003). The barbed glumes and the awns are instrumental in dispersal because they will work the seed into animal hair or clothing, facilitating transportation over long distances, and facilitate seed movement through litter (Stewart and Hull 1949).

Seed production and seed bank dynamics are important variables in the success of *B. tectorum* invasions. Each plant can produce 10-250 seeds, observed in unburned areas, and 960-6,000 seeds, observed in some burned areas (Young and Evans 1978). However, seed production is principally dependent on precipitation and the density of plants, where at lower densities more seeds and biomass are produced (Stewart and Hull 1949; Hulbert

1955; Young et al. 1969). Plants under grazing or moisture stress, that only grow 2.5 - 5 cm tall, can still produce seed (Stewart and Hull 1949). *B. tectorum* can also be infected by smuts such as *Ustilago bromivora* which attack the seeds, filling the caryopses with black hyphae and spores (Stewart and Hull 1949). Seed longevity is variable, often depending on precipitation. Smith et al. (2008) found that 96% of seeds germinated in the first year, 3.6% germinated the second year, and 0.4% germinated the third year after seed production, where additional seed input was suppressed. However, Young and Evans (1975) found that seed germination was higher under litter compared on bare soil, leading to decreased seed longevity because there were fewer seeds to germinate. This could indicate how the study by Smith et al. (2008) could have inflated estimates of seed longevity because of the reduced litter conditions present in their study with the removal of plants to avoid additional seed input. Even though seed longevity is typically not long, the number of seeds produced by each plant makes the species capable of invading or recovering rapidly in an area.

Safe sites, which include slight depressions and litter, are areas that will increase the potential for germination and seedling survival through increased seed-soil contact, more favorable temperature and moisture conditions, and possibly protection from consumption (Evans and Young 1984). *B. tectorum* germinates better under litter or in slight depressions than on bare soil (Evans and Young 1972). Evans and Young (1970) found that the number of *B. tectorum* plants established under litter was three times greater than on bare soil. Young et al. (1969) found that the majority of the current year's seed production was found in the litter layer, indicating that it acted as a collection point in addition to becoming an important location of safe sites. The seeds have more time to germinate or work into the soil, as well as take advantage of any moisture held above the soil (Young and Evans 1975). The majority of seed germination within the soil occurs in the top 2.5 cm, indicating the importance that the depth of seed placement has on survival (Wicks et al. 1971). In comparison to *B. tectorum's* preference for litter, other annual species such as Russian thistle, halogeton (*Halogeton glomeratus*), and mustards (*Sisymbrium altissimum*, *Descurania pinnata*) do well on bare soil, later depositing litter which allows *B. tectorum* to invade a system which was previously disturbed. Small seed sizes of many annual species increases seed-soil contact, while mustard seeds also produce a mucilaginous coat which increases soil contact and moisture retention (Evans and Young 1984). While a litter layer composed of *B. tectorum* and other annual species is beneficial for *B. tectorum* growth, it can act as a physical barrier for other species, in terms of the increased distance for the seedling shoot to reach sunlight and roots to reach the soil (Facelli and Pickett 1991).

B. tectorum has an average air-dried yield of 224 kg ha $^{-1}$; however, Uresk et al. (1979) found a range of biomass production from 132 to 328 kg ha $^{-1}$ over a 5- year period. It grows to a height of 30-50 cm in normal precipitation years, but can vary from 5-8 cm with low precipitation to greater than 60 cm in high precipitation years (Stewart and Hull 1949). Biomass production is a major factor facilitating the expansion of *B. tectorum* because it carries over from year to year, adding to the litterbed. Biomass retained by the system as litter can vary from a scattering of dead material to a mat 1.5 cm or thicker (Stewart and Hull 1949), which as stated previously, acts as a continuous fine fuel to increase fire frequency and severity. The other aspect of litter in connection with fire is that once litter buildups and fires begin, a fire cycle is started where fire encourages growth of *B. tectorum*, which then encourages fires (Rice 2005). Young and Allen (1997) indicated that without woody fuels to increase fire severity, grass fires in *B. tectorum* areas are typically not able to reduce the seed bank, which is held in the litter layer and top few cm of soil. However, recent studies have shown that fires in *B. tectorum*-dominated communities without woody fuels significantly reduce the litterbed and *B. tectorum* seed bank densities (Humphrey and Schupp 2001; Call et al. 2008).

Manipulation Practices

A common method for reducing undesirable species is the application of herbicides on rangelands and agricultural areas (Young et al. 1981; Peeper 1984). Many different herbicides have been used for *B. tectorum* control, such as the foliage applied herbicides paraquat, dalapon, quizalofop (Assure II), fluazifop-p-p-butyl, sethoxydim (Post), glyphosate (Roundup), and the soil active herbicides atrazine, imazapic (Plateau), and sulfometuron methyl (Oust) (Evans and Young 1984; Carpenter and Murray 1998).

Dalapon, a leaf contact herbicide, has been used to control weeds and to assist in seedbed preparation by reducing weed densities (Vallentine 1989). Kay (1963) studied the impact of dalapon on the invasive annual grass medusahead (*Taeniatherum caput-medusae*). He found that the best control was achieved during early growth stages at lower spray rates than would be required at later stages. However, some problems with dalapon are that it reduces desirable grasses, it allows undesirable annual forbs to increase due to its soil residual time (approximately 6 weeks) and it is no longer labeled for rangeland use (Vallentine 1989).

Paraquat is also a leaf contact herbicide that is adsorbed and deactivated when it comes in contact with soil (Kay and Owen 1970). The use of paraquat can drastically reduce *B. tectorum* plants; however, new individuals can establish from the seed bank within the same season (Evans et al. 1967). In other work, however, paraquat treatments exhibited weed control throughout the season (Kay and Owen 1970). The problem with this contact herbicide is that any plants or leaves that are not contacted by the spray are not affected. Paraquat can also increase annual forb abundance because litter sprayed with the herbicide decays in a month, revealing bare soil which is preferred by the forbs (Evans et al. 1974).

Many studies have been performed using the atrazine/fallow technique for controlling *B. tectorum*, often in connection with applications of 2,4-D or paraquat, and have been very successful (Evans and Young 1977; Young et al. 1969; Eckert et al. 1974). However, Young et al. (1969) found in one atrazine/fallow experiment that while current populations of *B. tectorum* were killed completely, there was little effect on the seeds in the seed banks, so in subsequent germination periods, there was no reduction of seedling survival. Kay and McKell (1963) found that the effectiveness of atrazine in controlling undesirable species was dependent on the amount of fall precipitation, which promoted germination and growth before cold temperatures in the winter. The atrazine/fallow treatment decreases undesirable species, allows soil moisture conservation, and the accumulation of nitrogen (Eckert et al. 1974). One problem with the use of atrazine is that if improper rates are used, there could be levels of herbicide residue remaining in the soil beyond the first year of the treatment (Eckert et al. 1974). Additionally, atrazine is no longer labeled for rangeland or non-crop use.

Growth regulator herbicides have also been studied to determine if they will reduce *B. tectorum*. If aminopyralid or picloram (typically broadleaf herbicides) are applied to grasses after the initiation of grass jointing during the development of reproductive parts, the resulting seed is underdeveloped and nongerminable (Rinella et al. 2010). Rinella et al. (2010) found that this type of herbicide application on Japanese brome (*Bromus japonicus*) resulted in a 95 percent reduction in germinable seed. The advantage of this type of herbicide would be reduced damage to perennial grasses and damage to any exotic forbs that may be present; however, it would also cause damage to any remnant perennial forbs.

Tebuthiuron has also been observed to decrease *B. tectorum* density, and could lead to an increase in perennial species from annual dominated sites (Blumenthal et al. 2006; Olson and Whitson 2002). However, the removal of shrubs increases the availability of resources which could lead to an increase of annual species if perennial species are not dense enough to use all the newly available resources (Olson and Whitson 2002).

Currently, the herbicide that is most often applied for *B. tectorum* control is imazapic, a soil active herbicide with an average persistence time in the soil of 120 days (Kyser et al. 2007). This persistence time results in continual control throughout the treatment period, so that mainly seedlings of *B. tectorum* are affected. It has also been shown that many established native perennial grasses and forbs are tolerant of imazapic; however, injury can occur to seeded perennial grasses depending on the timing or rate of application (Shinn and Thill 2004; Kyser et al. 2007; Sheley et al. 2007). Imazpic is often applied in the fall to impact germinating seeds, and because fall timings have been found to be more effective than spring applications (Sheley et al. 2007). One problem with imazapic is that if it contacts litter rather than soil, it can bind to the litter, reducing its effectiveness (Kyser et al. 2007).

Prescribed grazing has been commonly used to negatively impact undesirable species by using grazer preference to decrease one plant type so another can increase (Vallentine 1989; Young and Clements 2007). Often, this involves using cattle to increase shrubs because they prefer grasses, or goats or sheep to increase grasses because they focus on shrubs and forbs. The reduction of biomass or seed production through grazing can lead to reduced fitness and decreased ability to compete with surrounding plants as growth rate, size and density are decreased (Olsen and Richards 1989; Marsh 1990; Vallentine and Stevens 1994). Grazing can be used to suppress less desirable plants such as *B. tectorum*, which reduces moisture stress and increases the establishment and growth potential of other plants (Vallentine 1989). Intensive grazing can reduce *B. tectorum* seed input and the density of seed in the seed bank (Call et al. 2008). In a simulated grazing study, plants clipped at the boot stage and 2 weeks later had much lower seed production than unclipped plants (Hempy-Mayer and Pyke 2008). Grazing decreases buildup of fine fuel loads, reducing the fire potential of an area (Davison 1996). Intensive grazing has also been used as a preparatory step for revegetation of *B. tectorum*-dominated sites; however, repeated grazing is often required to reduce *B. tectorum* to a level where other species can be seeded and successfully establish (Vallentine and Stevens 1994; Mosley and Roselle 2006).

There are also some potential biological control agents that could be used to control *B. tectorum*, such as pathogens, smuts (head and chestnut blunt), and rhizobacterium (Meyer et al. 2008; Dooley and Beckstead 2010). A fungal pathogen (*Pyrenophora semeniperda*), blackfingers-of-death, kills slow germinating seed and reduces the seed bank, while a deleterious rhizobacterium (*Pseudomonas fluorescens*) can inhibit root elongation and seedling vigor; however, the two cannot be used together because of negative interactions (Dooley and Beckstead 2010). Currently, research using the bacteria, *Pseudomonas fluorescens*, which stunts the growth of *B. tectorum*, is ongoing [Reneé Schultheis (Anne Kennedy), personal communication, January 13, 2011]. While biological control has not been as widely used as other treatments, it may be a technique for temporarily reducing *B. tectorum* abundance when used alone or in conjunction with other control methods (Meyer et al. 2008).

Mechanical treatments such as plowing and heavy disking in the spring or fall have been used for controlling *B. tectorum*. During the 1930's and up to the present, one of the most common uses of mechanical treatments on rangelands has been connected to seeding efforts, i.e. the stump-jump-plow and brushland plows with rangeland drills (Young and McKenzie 1982). However, problems with using only mechanical methods are that while viable seeds can be buried too deep for germination, others can remain near the surface for subsequent germination, and the disturbance can open sites for additional weed colonization (Young et al. 1969).

Prescribed burning is an additional control method, especially for the reduction of litter (Carpenter and Murray 1998; DiTomaso et al. 2006). Early summer burning has been successful for thinning annual grasses such as *B. tectorum* and reducing seed input (Stewart and Hull 1949; Rasmussen 1994). Seed input can be decreased if fires occur before seed shatter so the seeds are still up in the fuelbed where they can be killed by the heat from the fire (Rasmussen 1994). However, problems with summer burning include damaging perennial grasses before they have matured, and control of fires. For these reasons, prescribed burns are more common in the fall (Wright 1974). Fire can increase the abundance of annual grasses by increasing the degree of disturbance. However, if seeds are remaining above the ground-line, annual grass abundance can be decreased by reducing the seed bank size, which allows other species the opportunity to reenter the system. Fire can return the area to bare soil, reducing the ability of *B. tectorum* to grow, while desirable species can be drill seeded into the prepared seedbed. However, fire coverage is not always complete, especially in areas without woody materials which help increase fire intensity, and small patches of litter are left with their seed banks intact (Young et al. 1976). Fires may not always create conditions that are detrimental to *B. tectorum* germination, because ash accumulation can be a nutrient-rich seedbed for seeds that were not damaged by the fire (Young et al. 1976).

Davies (2010) showed that the combination of prescribed fire and imazapic application had the greatest effect on reducing medusahead (*Taeniatherium caput-medusae*), an invasive annual grass, than either treatment alone. He also hypothesized that the long term effects of an herbicide-only application would not be significant, as annual grasses could recover even though there was a significant reduction with imazapic that was not seen with burning alone (Davies 2010).

After the advantage that *B. tectorum* gained with litter and seed production has been reduced through grazing, herbicide, burning, and/or mechanical treatments, seeding of desirable species is possible and is widely used (Stewart and Hull 1949; Vallentine 1989). Reintroduction of desirable species is often necessary in order to prevent reestablishment of non-native annual grasses and forbs because of annual species' ability to capitalize on resources made available by disturbances and the reduced competitive effects by perennial species (Prevey et al. 2010). However, unless competition by *B. tectorum* is reduced, seedings often fail (Platt and Jackman 1946; Davies 2010). For long periods of time the species that has been seeded most often is crested wheatgrass (*Agropyron cristatum*, *A. desertorum*), an introduced perennial grass (Young and Allen 1997). Crested wheatgrass is used because it has the ability to grow roots similar in form and seasonality to *B. tectorum*, utilizing moisture early in the season, which allows it to compete with *B. tectorum* (Stewart and Hull 1949; Harris and Wilson 1970). Crested wheatgrass is also tolerant of heavy spring grazing (Young et al. 1981). Crested wheatgrass seeding has been combined with mechanical treatments because it grows best when planted 3.8-5 cm deep; however, large soil disturbance can either bury annual seeds or just provide more areas of invadable disturbance (Stewart and Hull 1949; Young et al. 1969). Other species that have been successful in seeding treatments include 'Luna' pubescent wheatgrass (*Agropyron trichoporum*), 'Sodar' streambank wheatgrass (*Elymus lanceolatus*), and to a lesser extent, Russian wildrye (*Psathyrostachys juncea*) and Critana thickspike wheatgrass (*Elymus lanceolatus*) (Whitson and Koch 1998). Additional studies have looked at the interactions of native species, including bottlebrush squirreltail (*Elymus elymoides*), needle and thread grass (*Hesperostipa comata*), and rabbitbrush (*Chrysothanmus viscidiflorus*), with *B. tectorum* (Melgoza et al 1990; Humphrey and Schupp 2004). However, no matter what species are used, James and Svejcar (2010) have shown that it is important to use the correct seeding technology and understand the background characteristics of the area in order to have the best chance of establishment.

Managers and scientists have implemented many different treatments, alone and in combination, to improve rangelands. Some projects have involved selective grazing and patch burning in combination with herbicides, herbicides and perennial grass competition, prescribed fire and herbicide with mowing, and many combinations of seeding treatments (Whitson and Koch 1998; Cummings et al. 2007; Simmons et al. 2007; Davies 2010). However, when the management objectives were focused on treatments which typically involved killing a target species, they often failed (Krueger-Mangold et al. 2006). In contrast, Evans et al. (1970) experimented with paraquat and seeding to reduce the abundance of *B. tectorum*, focusing on changes in ecosystem attributes, not only killing *B. tectorum*. Successional management is this movement from a focus on treatments, to what those treatments do cumulatively to systems as a whole. Sheley et al. (1996) proposed a management framework based upon the successional theory of Pickett et al. (1987). Those ideas and others have developed into approaches such as the Ecologically-Based Invasive Plant Management (EBIPM) framework, where managers focus on the system and all its processes to make decisions based on ecological principles, and not just focusing on killing individual plants (Krueger-Mangold et al. 2006; Sheley et al. 2010).

Ecologically-Based Management

To have an impact on an ecosystem, it is necessary to understand what processes shape and maintain it. Many areas in the Great Basin are currently dominated by cheatgrass. The historical natural community of many of these areas was composed of shrubs, typically *Artemisia* spp., with understories of grasses and forbs. The historical state of sage-steppe cycled between dominance of the shrubs and dominance of the understory species through various feedback mechanisms that reinforced ecosystem resilience (Briske et al. 2008). One of the major mechanisms was the fire return interval of approximately 50 to 75 years, which reduced the shrub overstory enough that an understory was able to grow, and allowed the shrubs enough time to recover after the fires. Ecologically unsustainable livestock grazing practices, and the introduction of the annual *B. tectorum* caused changes in the ecological processes of the areas, introducing the potential for change.

The work of Krueger-Mangold et al. (2006) shows how managers can use treatments to manipulate ecosystems by focusing on how ecological processes are affected by specific treatments. This is described in the successional management framework, in which causes of succession are affected by the processes and components inherent within ecosystems, which are in turn altered by modifying factors (Table 1) (Sheley et al. 1996; Krueger-Mangold et al. 2006). The three causes of succession are: site availability, species availability, and species performance. The causes of succession are influenced by focusing management on the modifying factors of individual processes or components. As processes or components are changed, a transition to a different state is possible.

Table 1. Causes of succession, processes and components, and modifying factors (Krueger-Mangold et al. 2006).

Causes of Succession	Processes and Components	Modifying factors
Site availability	Disturbance	Size, severity, time intervals, patchiness, predisturbance history
Species availability	Dispersal	Dispersal mechanisms and landscape features
	Propagule pool	Land use, disturbance interval, species life history
Species performance	Resource supply	Soil, topology, climate, site history, microbes, litter retention
	Ecophysiology	Germination requirements, assimilation rates, growth rates, genetic differentiation
	Life history	Allocation, reproduction timing and degree
	Stress	Climate, site-history, prior occupants, herbivory, natural enemies
	Interference	Competition, herbivory, allelopathy, resource availability, predators

Site availability, such as the loss or gain of safe sites particular to different species, is often altered with disturbances such as fire. Fires often reduce a litter covered area to bare soil and kills perennials that are unable to resprout. Water run-off can dramatically alter soil surface characteristics, reducing the depressions and cracks necessary for seed survival. All of this returns the potential of the area to an early successional stage, of which *B. tectorum* becomes dominant. *B. tectorum* has the ability to use water and nutrients earlier in the season than other native species, allowing it to take advantage of the resources released by the fire because previous individuals that used the water and nutrients are either dead or trying to recover, similar to how *B. tectorum* utilizes resources before other species can earlier in the year because of its early growth (Harris 1967; Melgoza et al.1990). These characteristics allow *B. tectorum* to invade and spread in a community that has been altered by disturbance. *B. tectorum* dominance then changes the basic processes of the area, as when fires become more frequent, changing the time intervals, and often the size and intensity, all factors of disturbance. However, fire can alter the site availability in an annual grass-dominated system, which can improve the chances for establishment of seeded species by reducing the litter cover composed of annual grasses. This reduces safe sites for annual species and opens sites for seeded species because they do not have to work through that litter layer (Davies 2010). Davies (2010) also showed that by reducing the litter layer and altering site availability, the addition of a herbicide treatment was more effective because of the increased soil contact.

Species availability is influenced by two processes and components, dispersal and the propagule pool. Altering seed dispersal is one way to affect the ability of the species to spread and grow. If a plant can be manipulated to disperse few propagules and is an annual, the majority of the individuals for the next year have to come from the seed bank. This reduces the ability of the population to survive for multiple years if seed reduction continues to occur. It is also possible to influence the species availability of desirable species through additions of seeds to the environment, altering their propagule pool numbers. Davies et al. (2010) designed a study to determine if planting a strip of desirable, competitive vegetation could be used to prevent the spread of an invasive grass. Their study showed that medusahead spread was slowed by crested wheatgrass plants physically stopping propagule dispersal and because of an increased distance from invaded sites to other plant communities. However, they stated that more research was needed to actually determine if the invasion front was actually slowed and if biotic resistance of the protected community was increased (Davies et al. 2010).

The ability of a plant to respond positively to biotic and abiotic conditions is related to species performance, the third cause of succession. Some of the factors that could be of use in controlling *B. tectorum*-dominated systems are associated with life history traits (i.e., being a winter annual), which if used for manipulation and species choices for seeding, can affect two of the other factors, stress and interference. This can occur by seeding a species that can compete with the annual for resources, reducing its prevalence across the landscape. The success of a species in a community is dependent on its ability to compete with or tolerate other individual plants. It is possible to affect species performance by adding other agents, such as cattle, which will impact the growth and development of plants through herbivory. Fire is an important factor in species performance because of the effect it has on the life cycle of a plant. Perennial species could be adversely affected by fire if it occurs before seed maturity is reached. Additionally, plants may be negatively impacted by fire or herbivory because of the need to expend resources to regrow damaged vegetative and reproductive structures. *B. tectorum* is affected by fire in many ways, both positively through increased production, as it has the opportunity to dominate a community, and negatively because the area is returned to bare soil, which is not as desirable as a litter bed for establishment.

Sheley et al. (2006) used the successional management framework to identify treatments that would have an impact on factors that would alter ecosystem structure and function in an ephemeral pothole wetland area of Montana. The major habitat type, rough fescue/ bluebunch wheatgrass (*Festuca campestris/ Pseudoroegneria spicata*), had become dominated by the invasive forbs spotted knapweed (*Centaurea maculosa*) and sulfur cinqefoil (*Potentilla recta*). Different herbicides were used to alter species performance, different seeding methods to alter site availability (amount of disturbance), various seeding rates to alter species availability, and different cover crops to alter the species performance via competition for resources. All of these treatments have been used in weed control; however, the focus of their work was on the modifying factors which influenced change rather than on treating the weeds alone. Sheley et al. (2006) noted that it may take time and recurrent treatments to achieve change in all necessary processes and components of the target area, and the ability to use successional theory is limited by how processes influence plant dynamics. To date, successional weed management has not been widely applied because it has not been linked to other successional models (Kruger-Mangold et al. 2006). Incorporating other successional models, such as state-and-transition models, will widen the possibilities for ecologically-based management of invasive weeds such as *B. tectorum*.

An important aspect of each ecosystem is the potential for change and what alterations in the ecosystem happen because of change. This interaction has been described historically as a linear process leading to a climax community, multiple stable states, and more recently by a state-and-transition model (Stringham et al. 2003; Briske et al. 2005). A state-and-transition model describes what the structure of the ecosystem is currently, what it may have been in the past, and what may potentially occur in the future, as well as the dynamics inherent in the ecosystem. State-and-transition models are comprised of various states, each containing a set of similar community phases connected by reversible transitions requiring little or no input (Fig. 1). When a state is relatively stable it can exist as one or more different community phases. Between states are transitions that are often irreversible without significant input and management to restore ecological processes, where the trajectory possibilities to different states are determined by natural events or management. Inherent in the irreversible transitions are thresholds, points where the ecological processes maintaining a system are changed "beyond the point of self repair" because of a set of interacting components (Stringham et al. 2003; Briske et al. 2005). When ecosystems are manipulated to create change, it is possible that the entire ecosystem is affected and those manipulation treatments become triggers with the potential to move the ecosystem to a different state. The variables that caused sage-steppe ecosystems to move across a threshold to *B. tectorum*-dominated states were non-sustainable grazing practices and the introduction of invasive species. However, management treatments are implemented in order to act as positive triggers which will move ecosystems to a more desirable state. Thresholds are difficult to reverse because processes have changed, not just species composition. In the example of sage-steppe ecosystems becoming dominated by *B. tectorum*, there are concerns with erosion after fires, causing changes to soil structure, and resources being released from the area. Manipulating processes is difficult and costly in terms of resources and time. However, by better understanding site specific state-and-transition model pathways, as well as the triggers that move ecosystems from one state to another, it is possible to identify management practices that can be used to move an ecosystem from one community phase to a different community phase or state (Briske et al. 2006; Krueger-Mangold et al. 2006).

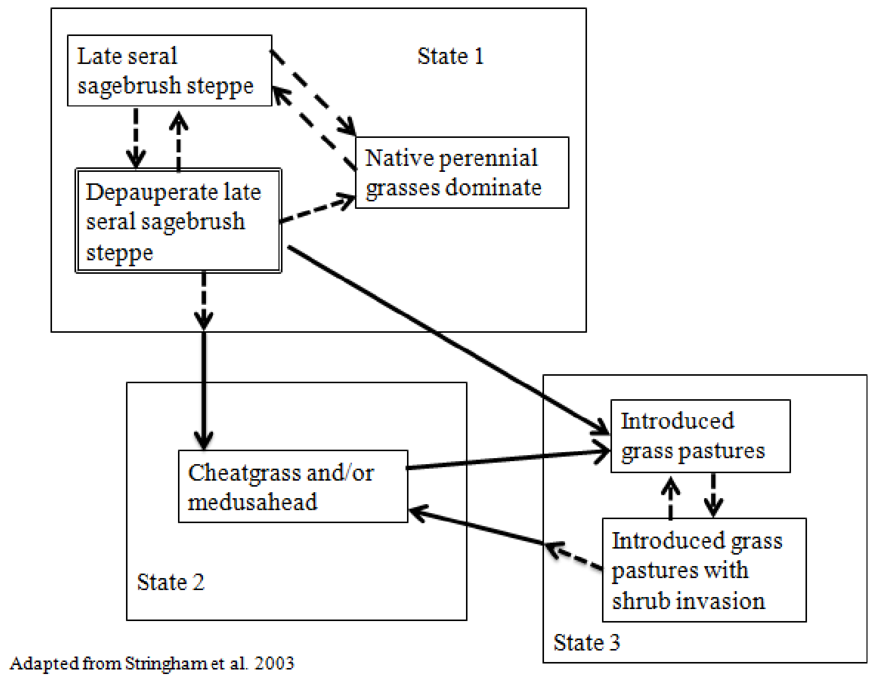

Figure 1. Example of a representative state-and-transition model for a sagebrush-steppe ecosystem (Stringham et al. 2003). The dashed lines are easily crossed transitions, the solid lines are difficult or nonreversible transitions. State 3 is where changes were made to state 1 either because the land manager desired more forage species or the potential for a threshold being crossed in the depauperate sagebrush community phase was identified, and rather than allowing a potential conversion to undesirable annual grasses, the phase was converted to a managed grass pasture in a different state.

If managers can understand the linkages between management practices, ecological processes and vegetation dynamics, then they will be able to utilize an ecologically based invasive plant management (EBIPM) framework or model to assist with decision-making (Sheley et al. 2010). Sheley and co-workers (2010) have created a more useful model by linking the processes and mechanism directing plant community change to ecological principles on which managers can base their decisions. The comprehensive model also improves decision-making with the inclusion of the Rangeland Health Assessment protocol (Pellant et al. 2005) to identify ecological processes in need of repair, and adaptive management features to promote flexibility and learning during management (Reever-Morghan et al. 2006).

Objectives

The goal of this research project is to convert a *B. tectorum*-dominated community to a healthy, functionally-diverse community that is resistant to weed invasion and repeated wildfires, and meets multiple land use objectives. Specific objectives are to: 1) use the EBIPM framework to design a series of manipulation treatments (grazing, prescribed burning, herbicide, and seeding) to suppress *B. tectorum* and promote the establishment of desirable species; 2) evaluate the effectiveness of these manipulation treatments in a large-scale management setting; and 3) contribute findings from Utah research sites to the area-wide EBIPM program.

The treatments included combinations of mowing to reduce biomass and seed input, fall prescribed burning to reduce litter and increase site availability for other species, and fall-applied imazapic herbicide to reduce *B. tectorum* seedling growth and survival. Following these treatments, a mix of native and introduced species was drill seeded into the area to introduce desired species. The effectiveness of the treatments in altering the causes of succession and creating a weed resistant community was evaluated.

METHODS

Site description

The study area is located in northwestern Utah (Box Elder County), near the town of Park Valley, in the Fisher Creek watershed. The study sites are on two pastures owned by different ranchers south of Highway 30. The upper site is at 41.77307 degrees latitude and -113.288012 degrees longitude, covering 50 ha, and the lower site is at 41.758986 degrees latitude and -113.268614 degrees longitude, covering 80 ha, approximately 1 km south-east of the upper site. Both sites had been moderately grazed by cattle for at least the past 30 years. They had also burned four times since the late 1980’s, most recently in 1999 and 2004. The area is categorized as a salt desert shrub system composed of semidesert loam (Wyoming big sagebrush) and semidesert alkali loam (black greasewood) ecological sites (Soil Survey Staff 2008). Common species in these ecological sites are Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), saltbush (*Atriplex canescens*), shadscale (*Atriplex confertifolia*), rabbitbrush (*Chrysothamnus viscidiflorus*), Sandberg’s bluegrass (*Poa secunda*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and black greasewood (*Sarcobatus vermiculatus*), where composition is typically 35-55% perennial grasses, 5-10% forbs, and 40-55% shrubs (Soil Survey Staff 2008). With fire, sagebrush decreases while rabbitbrush, bluegrass, greasewood, and annual species increase. At the time of treatment initiation, the study sites were dominated by cheatgrass (*Bromus tectorum*), with other annual nonnative species such as tumble mustard (*Sisymbrium altissimum*), tansymustard (*Descurania pinnata*), Russian thistle (*Salsola tragus*), and some residual native species such as Sandberg’s bluegrass and bottlebrush squirreltail (*Elymus elymoides*). The area receives approximately 280 mm annual precipitation, of which 74 mm comes as snowfall from November through March. The mean annual maximum temperature is 14.9° C and the mean minimum temperature is 1.1° C, with a freeze-free period of 100-150 days (WRCC, Utah Climate Data 2008; Soil Survey Staff 2008). The soils on both sites are classified in the Kunzler-Lembos association, with 1 - 5 % slopes. The soils are moderate to very deep (down to 1.5 m), and well drained. The soils have a sodium adsorption ratio (SAR) rating of 12.6, a pH of 8.5, a calcium carbonate ($CaCO^3$) rating of 3%, an electrical conductivity (EC) of 6.2 millimhos / cm, and a cation-exchange capacity (CEC) of 12.6 meg/100g (Soil Survey Staff 2008). The soil chemistry indicates that plants in this area may have problems with salinity as well as aridity.

Treatments

Cheatgrass manipulation treatments included spring intensive grazing (which was changed to localized mowing), fall prescribed burning, and fall applied herbicide, with each treatment applied alone and in combinations, i.e. herbicide, fire, and grazing (mowing) alone, herbicide plus grazing (mowing), herbicide plus fire, fire plus grazing (mowing), grazing (mowing) plus fire plus herbicide, and a control. Each treatment plot was also seeded with a mixture of perennial grasses and a forb. Each of the treatment plots is large scale, approximately 8-15 ha in size in order to meet the objective of a demonstration study. The study sites were surrounded by a 4-strand barbed-wire fence with electric interior fence to divide individual plots for grazing.

Prescribed grazing with cattle was implemented when *B. tectorum* was in the boot stage in late May through late June 2009. During grazing events, cattle were to remove 70-80% of live aboveground biomass. Because of untrained cattle, areas too large for the number of available cattle, and low voltage on the electric fencing, the grazing treatment was abandoned. Mowing was performed in late June to simulate grazing, using a gas push mower with the blade set 5 cm above the soil, and occurred on half of the 3 x 3 m microplots randomly distributed across each grazing treatment plot (112 total mowed). Mowed biomass was collected in a bag and removed from the site. The grazing treatment was planned to allow us to impact site availability (litter for safe sites) and species availability (seed production) for *B. tectorum*.

Prescribed burning occurred on 4 November 2009. It was started and controlled by the Box Elder County fire crew using drip and propane torches. Weather parameters were not measured at the sites during the burns. The nearest comprehensive weather station (Blue Creek, UT, NRCS SCAN site; ~70 km east of study sites) recorded air temperature at 14-17 ^{o}C, average wind speed at 9.6 km/hr, and relative humidity at 23-28%. The fire treatment was planned to reduce site availability (litter for safe sites) and species availability (seeds suspended in litter) for *B. tectorum*.

Imazapic herbicide was applied at a rate of 71.6 g active ingredient (ai)/ha [296 mL mixed in 65 L of water] on 18 November 2009 using a fixed wing airplane. The herbicide treatment was planned to reduce species performance of *B. tectorum* by reducing survival of newly germinating plants.

All treatment plots were drill seeded with a diverse seed mix containing native and introduced perennial grasses and an introduced forb between 8 and 15 December

2009, after the other treatments were completed. The seed mix included: ‘Hycrest’ crested wheatgrass (*Agropyron cristatum x A. desertorum*), ‘Bozoisky’ Russian wildrye (*Psathrostachys juncea*), ‘Bannock’ thickspike wheatgrass (*Elymus lanceolatus*), ‘Anatone’ bluebunch wheatgrass (*Psuedoroegneria spicata*), ‘Vavilov’ Siberian wheatgrass (*Agropyron fragile*), and ‘Ladak’ alfalfa (*Medicago sativa*). The seed mix was applied at a higher rate than is recommended for traditional rehabilitation practices because of potential seedling injury related to imazapic application (Shinn and Thill 2004). Due to mechanical/operator error, the upper site was seeded at a rate of 7 kg/ha pure live seed (PLS) while the lower site was seeded at 14 kg/ha PLS. Because only one seed mix was used, the seeding was applied across the entire area, including the control plots.

Measurements

Resident species aboveground biomass (mid-July), density (early to mid-June), cover (mid-July), seed production (early to mid-June), and plant height (early to mid-June) were measured in a Daubenmire frame (20 x 50 cm) at 30 randomly located sample points in each treatment plot (macroplot) in 2009 and 2010. Vegetation, litter, bare ground and rock cover was determined by ocular estimate for the entire Daubenmire frame. Aboveground biomass within the frame was clipped at the soil surface, separated by functional group (annual grass, annual forb, perennial grass, perennial forb, and perennial shrub), dried at 60°C for 72 hours, and weighed. At each point, measurements were taken at different locations in a 3 x 3 m hexagon to avoid destructive sampling effects. Because of the grazing treatment failure, a mowing treatment was implemented in early-July 2009 within randomly selected microplots. Seed bank density, litter depth, and seeded species density were measured in all microplots, mowed and unmowed, in November 2009 and May 2010. Seeded species density was also measured at each of the 30 sample points in each treatment plot in May and July 2010. A larger frame (1.2 m x 1.3 m), covering three drill rows, was used to measure seeded species density (Vogel and Masters 2001).

B. tectorum seed production was measured to quantify the effectiveness of the mowing and fire treatments in reducing seed input. Before seed shatter, early- to mid-June 2009 and 2010, any stems with inflorescences were clipped within a 20 x 50 cm Daubenmire frame at 30 sample points in each treatment plot and stored in paper pages to air dry. The number of spikelets per stem and the number of florets per spikelet were counted.

Seed banks were quantified in prescribed fire, mowing plus fire, and control treatments in October 2009, and in all treatments in May 2010. The sampling occurred only within microplots, due to the inclusion of the mowing treatment. For each sampling period, four Daubenmire frames were placed approximately 1.5 m inside the edge of the microplot borders (one on each side) and samples of litter and soil were collected from the corners of each Daubenmire frame, for a total of 16 litter and 16 soil samples within a microplot. Soil cores (5 cm diameter, 4 cm depth) were taken after the associated litter (5 cm diameter) was collected from the soil surface. The samples were combined into a composite soil sample and a composite litter sample for each microplot. A subsample (5 cm diameter, 4 cm depth) was taken from each composite soil sample in order to reduce the amount of soil needed for seedling emergence counts in the greenhouse (see below).

This was not done with the composite litter samples because of the varying amounts of litter collected in each microplot. Soil samples and litter samples were cold stratified (2-3 °C) for 3-4 months before being spread, in the greenhouse, on sterilized-sand-filled trays (2.5 cm deep), with wooden dividers between samples (27 x 52 x 6 cm trays, dividers 10.6 cm apart, or twice the distance for large litter samples to retain even depths). Total depth of the sample was approximately 1-2 cm for a total of 4-5 cm within each tray. Soil samples were consistently 1.5 cm deep above the sand. For the first set of samples (February-May 2010), commercial greenhouse lights with 400 watt sodium bulbs were placed 1.2 m above the trays because of the shorter winter days, and were set to be on for 8 hours. The amount of light given off by the lamps was 100-150 µmol/m^2s of photosynthetically active radiation. The trays received water through an overhead automated sprinkling system (360° fogger-mister nozzles, 0.9 m above the trays) delivering between 150 - 200 mL throughout the day. The temperature was set for a high of 22 °C (day) and a low of 18 °C (night). Because the second set of samples was placed into the greenhouse when there was more daylight (September-December 2010), supplemental lighting was not used. The trays were watered for approximately 6 weeks while seedling emergence was measured. At the beginning of the watering period, seedlings emerging from samples were identified, counted and removed every day until the germination slowed down. When germination was slow enough that all samples could be counted in a few hours, the sampling was performed 2 - 3 times per week. Samples were allowed to dry for 2-3 weeks before they were rewetted and monitored for the emergence of seedlings from dormant seeds for an additional 3-6 weeks.

Experimental Design

Treatments at each study site were arranged in a split-split plot design with two replications. The whole plot was the prescribed fire treatment (fire or no fire). The whole plot was split with the herbicide treatment (herbicide or no herbicide), which was applied in strips. The split-split plot was the mowing treatment (mowing or no mowing), which was located within individual microplots across the other treatments (Fig. 2).

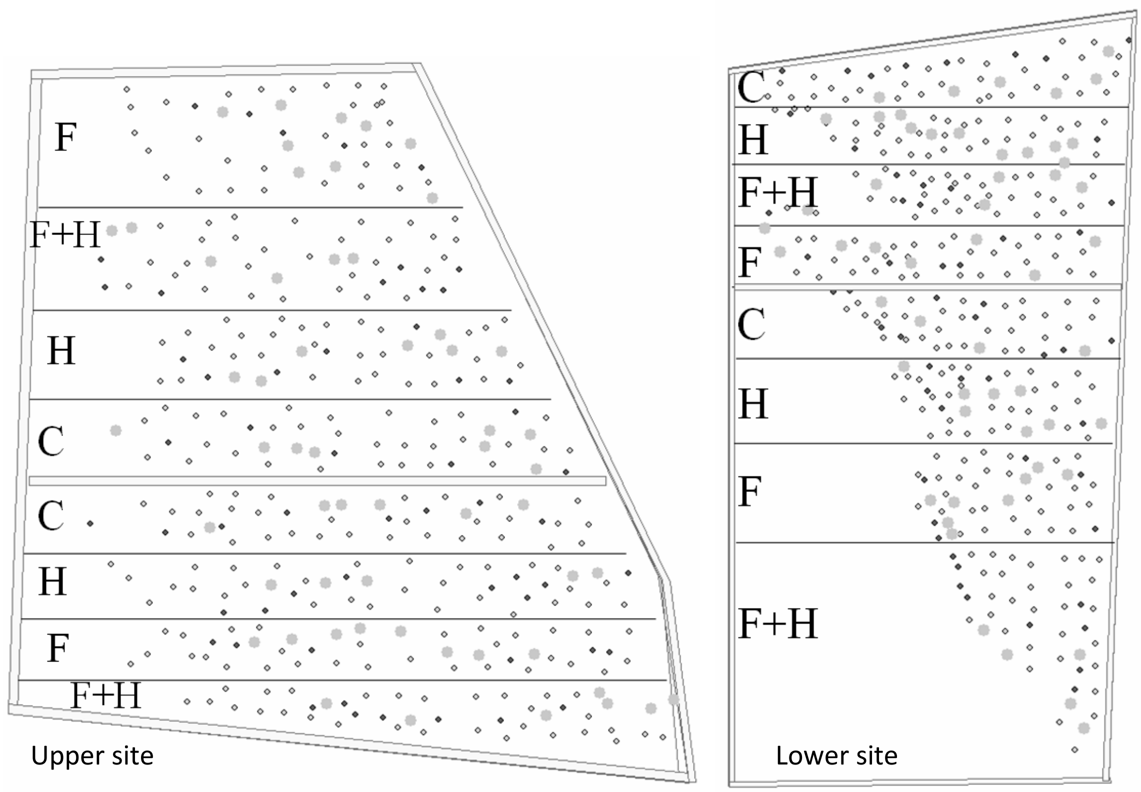

Figure 2. Treatment plot layout: F= fire, H = herbicide, F*H= fire and herbicide, and C=control. Larger grey circles are mowed/clipped microplots, black circles are unmowed microplots, and smaller grey circles are sampling locations within the macroplots.

For data collected at the macroplot level, the effects of fire and herbicide on the difference between 2009 and 2010 in resident plant responses (cover, biomass, density, seed production) were assessed using an analysis of variance (ANOVA) for a 2-way factorial in a split-plot design with whole plots in blocks (sites) and subsamples. Whole plot units were site halves where the factor was fire with two levels (fire and no fire). Subplot units were site quarters nested within site halves where the factor was herbicide with two levels (herbicide application or not). Daubenmire frame samples within site quarters were subsamples. For data collected at the microplot level, the effects of fire, herbicide and mowing were assessed using an ANOVA for a 3-way factorial in a split-split-plot design with whole plots in blocks (sites). Whole plot units were site halves where the factor was fire. Subplot units were site quarters nested within site halves where the factor was herbicide. Sub-subplot units were nested within site quarters where the factor was mowing (mowed and not mowed). In addition to responses of selected individual species, responses by functional group (annual forbs and perennial grasses, obtained as sums over pertinent species) were analyzed for each site. When data included pre-treatment and post-treatment data, the difference between the years was taken in order to eliminate the naturally occurring variability. Data were transformed as needed to better meet assumptions of normality and homogeneity of variance (Tables A1 and A2, Appendix A). Data analyses were computed using the GLIMMIX procedure in SAS/STAT software for Windows, version 9.2. All data are presented in figures and tables as untransformed main effects and interaction means. For an experimental factor such as fire, the fire main effect mean is the average (pooling) of the fire/no-herbicide simple effect mean and the fire/herbicide simple effect mean, and the no-fire main effect mean is the average (pooling) of the no-fire/no-herbicide simple effect mean and the no-fire/herbicide simple effect mean. Differences between means were considered significant if $P \leq 0.10$. Because of the multi-way factorial in a mixed model, patterns of significance shown by the model may not be apparent within figures because the standard errors do not mimic the model. All ANOVA test results with summary tables of significant effects are shown in the tables found in Appendix A and are not referred to because significant p-values are provided in the text. Tables A66 and A67 within Appendix A summarize which treatments were found to have significant results for all measurements at both sites. All tables and figures referred to in the Results section are presented after the Management Implications section and show untransformed data, and where means are displayed, standard deviations are also given.

RESULTS

Site Characteristics

Environmental conditions were considered similar at both sites because they are only 1.5 km apart. Precipitation data (Fig. 3) for 2008 through 2010 were from the Rosette, UT monitoring station (http://climate.usurf.usu.edu/products/data.php), approximately 10 km from the research sites. The year before treatments were implemented, 2008, there was below average precipitation, which could have caused decreased vegetative growth and seed production. Above average precipitation in 2009 occurred throughout much of the growing season with a strong peak in June, delaying the maturation and senescence of *B. tectorum* until late-July. The 2010 precipitation regime was similar to the long-term average.

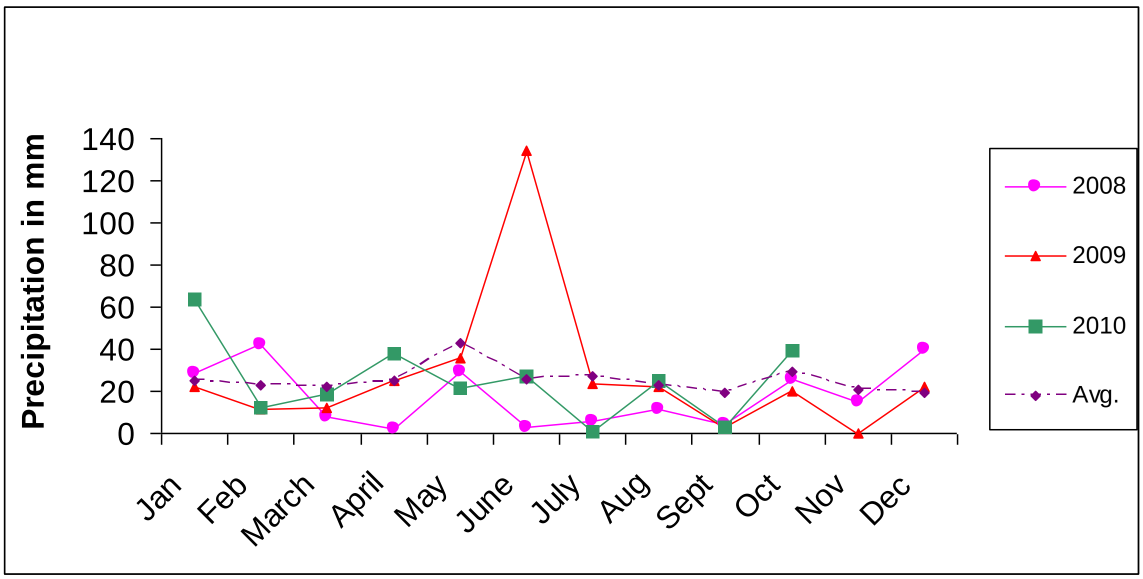

Figure 3. Monthly precipitation for Rosette, UT, 2008-2010. Data shown includes the long-term average precipitation as well as precipitation for the year before data collection (2008) and the two years of the study (2009 and 2010).

Temperature can also influence plant response to manipulation treatments. Monthly mean temperature data were taken from the same station as the precipitation data, and showed that for the two years of data collection (2009 and 2010), temperatures were lower than the long-term average (Fig. 4). These data indicate that for both years, precipitation and temperatures were favorable for resident and seeded species growth.

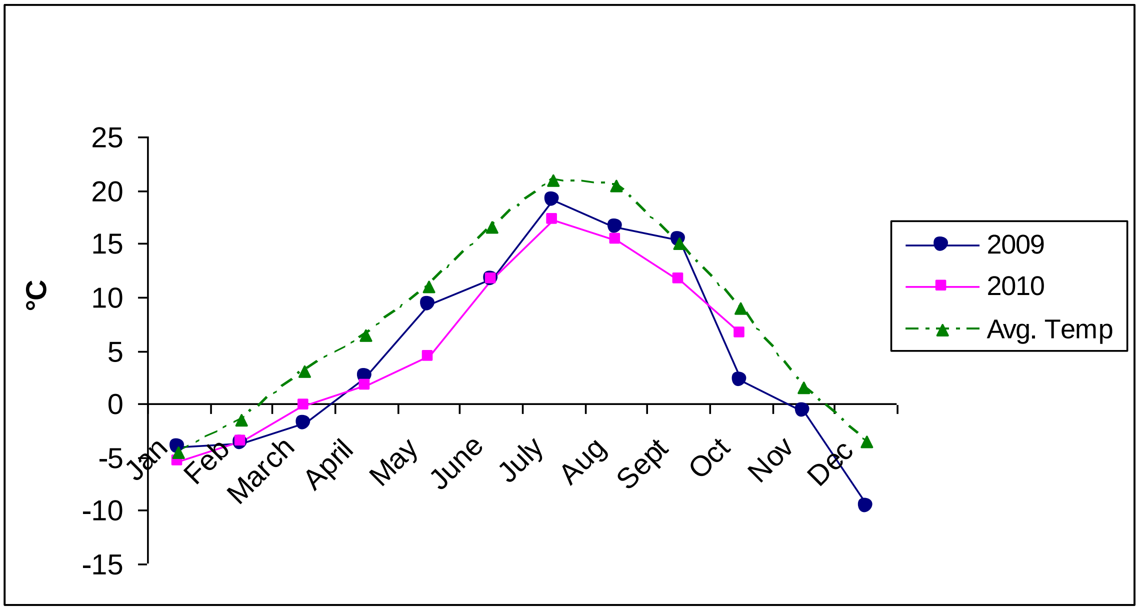

Figure 4. Average monthly temperature for Rosette UT, 2009 and 2010. Data shown includes the long-term average temperature as well as temperature for the two years of the study.

Treatment responses are based in large part on the pre-treatment vegetation and site conditions. Pre-treatment (2009) cover values for plant functional groups, litter, bare ground and rock differed between the upper and lower sites. The upper site (Fig. 5) had more *B. tectorum* and bare ground cover, and less litter and annual forb cover than the lower site (Fig. 6). The annual forb functional group mainly included *Sisymbrium altissimum*, *Salsola tragus*, and *Kochia scoparia*, with trace amounts of bur buttercup (*Ceratocephala testiculatus*), halogeton (*Halogeton glomeratus*), and *Descurania* *pinnata*. The perennial grass functional group included *Poa secunda*, *Elymus elymoides*, *Agropyron cristatum*, and trace amounts of Indian ricegrass (*Achnatherum hymenoides*). Plant species found at the site are listed in Appendix B.

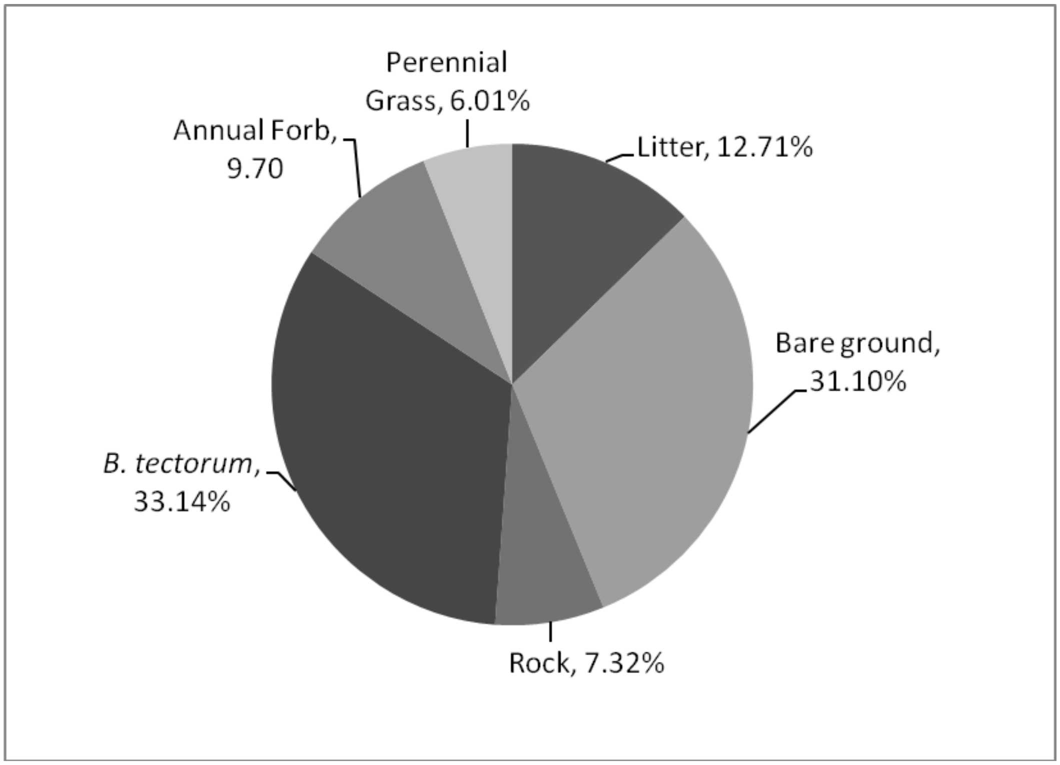

Figure 5. Mean percent cover for vegetation, litter, soil and rock components at the upper site, pre-treatment (2009).

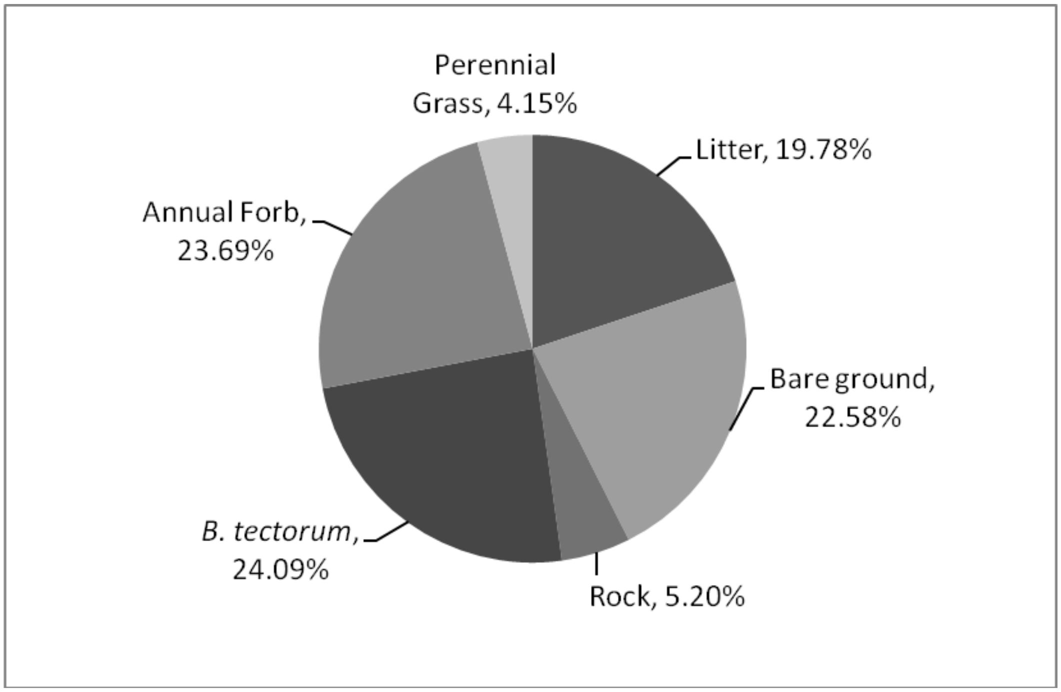

Figure 6. Mean percent cover for vegetation, litter, soil and rock components at the lower site, pre-treatment (2009).

Litter and Bare Ground

Litter cover increased in all treatments between 2009 and 2010 at both sites (Table 2); however, at the upper site the increase was significantly lower (P=0.0683) for fire than no-fire treatments. Treatment effects were non-significant at the lower site.

Depth of litter, comprised primarily of *B. tectorum*, was measured in conjunction with the collection of litter and soil seed bank samples. In fall 2009, litter depth was significantly reduced by the mowing treatment (P=0.0416) at the upper site, and by the fire treatment (P=0.0673) at the lower site. Fire had minimal impact on litter depth at the upper site, as did mowing at the lower site (Table 3). In spring 2010, mowing as a main effect was significant in reducing the depth of the litter without the presence of the fire or herbicide treatments at the upper site (Table 3). The magnitude of the mowing effect was determined by the fire and herbicide treatments, as indicated by the significant fire*herbicide*mow interaction (P=0.0744). However, without herbicide the presence or absence of mowing had no effect when combined with the fire treatment (Fig. 7). At the lower site, the fire treatment also showed a significant effect (P=0.0260), illustrating the trend that occurred with greater reduction of litter depth when fire was combined with other treatments.

Table 2. Significant (P<0.1) main-effect means (± 1 SE) for the differences between pre-treatment (2009) and post-treatment (2010) *B. tectorum* attributes for the upper and lower sites. Significant means in bold.

Attribute	No Fire	Fire	No Herbicide	Herbicide
Upper Site				
Litter cover (%)	**20.6 (5.4)**	**10.6 (1.8)**	12.5 (4.0)	18.7 (3.2)
Bare ground cover (%)	-8.8 (2.1)	12 (0.15)	**-5.7 (1.7)**	**8.8 (0.5)**
B. tectorum cover (%)	-7.5 (0.51)	-18.2 (8.4)	**-2.5 (0.9)**	**-23.3 (9.8)**
B. tectorum density (plants/m^2)	**1489 (118)**	**664 (56)**	**1428 (107)**	**728 (68)**
B. tectorum biomass (grams/m^2)	-117 (7)	-111 (7)	**-95 (6)**	**-134 (8)**
B. tectorum seed production (seed/m^2)	29394 (26702)	26133 (22643)	**54291 (43270)**	**1235 (6074)**
Lower Site				
Litter cover (%)	15.2 (1.7)	9.7 (0.9)	8.6 (1.3)	16.3 (2.1)
Bare ground cover (%)	**4.2 (2.1)**	**3.3 (2.7)**	0.7 (1.3)	6.8 (0.7)
B. tectorum cover (%)	1.5 (1.5)	-0.15 (4.9)	6.2 (3)	-4.9 (6.4)
B. tectorum density (plants/m^2)	1563 (109)	538 (45)	1270 (90)	831 (64)
B. tectorum biomass (grams/m^2)	-40 (3)	-49 (5)	**-33 (2)**	**-55 (5)**
B. tectorum seed production (seed/m^2)	9880 (3047)	10254 (4461)	**31467 (18016)**	**-11333 (10507)**

Table 3. Significant (P<0.1) main-effect means (± 1 SE) for litter depth (mm) and *B. tectorum* seed bank (plants / m^2) in 2009 and 2010 for the upper and lower sites. Significant means in bold.

Attribute	No Fire	Fire	No Mow	Mow	No Herbicide	Herbicide
Upper Site						
Litter depth '09	1.95 (0.18)	1.86 (0.14)	**1.68 (0.20)**	**0.67 (0.12)**	-	-
Litter depth '10	0.96 (0.12)	0.89 (0.12)	**1.09 (0.14)**	**0.75 (0.09)**	0.81 (0.1)	1.0 (0.14)
Litter seed bank '09	**19745 (11130)**	**9202 (5669)**	19067 (5669)	9879 (4172)	-	-
Soil seed bank '09	1257 (1007)	119 (28)	1127 (934)	249 (101)	-	-
Litter seed bank '10	5790 (1340)	2201 (486)	**5827 (1248)**	**2165 (578)**	3311 (867)	4681 (959)
Soil seed bank '10	**235 (46)**	**139 (34)**	**241 (48)**	**133 (33)**	236 (46)	138 (34)
Lower Site						
Litter depth '09	**4.12 (0.32)**	**2.89 (0.15)**	1.65 (0.23)	0.97 (0.23)	-	-
Litter depth '10	**1.54 (0.17)**	**0.64 (0.07)**	1.10 (0.13)	1.08 (0.12)	1.04 (0.12)	1.13 (0.12)
Litter seed bank '09	**26576 (11615)**	**7542 (5523)**	16427 (9177)	17691 (7961)	-	-
Soil seed bank '09	**1155 (667)**	**214 (80)**	328 (43)	1040 (705)	-	-
Litter seed bank '10	**7390 (1636)**	**895 (217)**	3504 (493)	4781 (1360)	4651 (1306)	3634 (547)
Soil seed bank '10	**279 (46)**	**71 (22)**	**187 (42)**	**164 (26)**	204 (40)	146 (28)

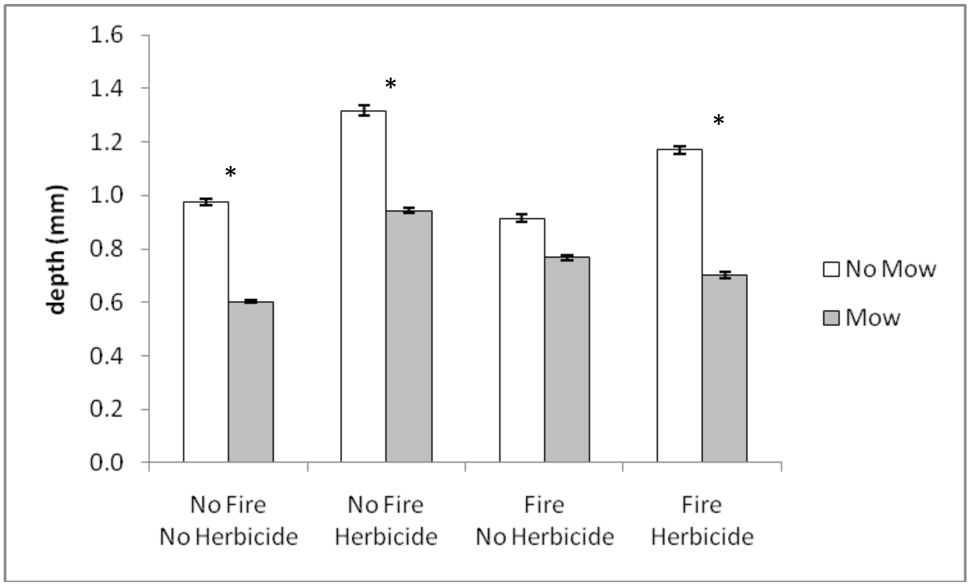

Figure 7. Mean (± 1 SE) litter depth at the upper site sampled May 2010. Asterisks indicate significant differences between no mow and mow treatments within a fire * herbicide treatment combination.

Bare ground generally increased for manipulation treatments between 2009 and 2010 at both sites (Table 2). At the upper site, there was a significant change in bare ground (P=0.0264) with the herbicide treatment. At the lower site there was a significant difference in bare ground (P=0.0793) with the fire*herbicide treatment. There were no significant differences in the increase in bare ground cover between the herbicide and no herbicide treatments without fire (Fig. 8). With both fire and herbicide, the amount of bare ground cover increased, while without herbicide there was a decrease (Fig. 8).

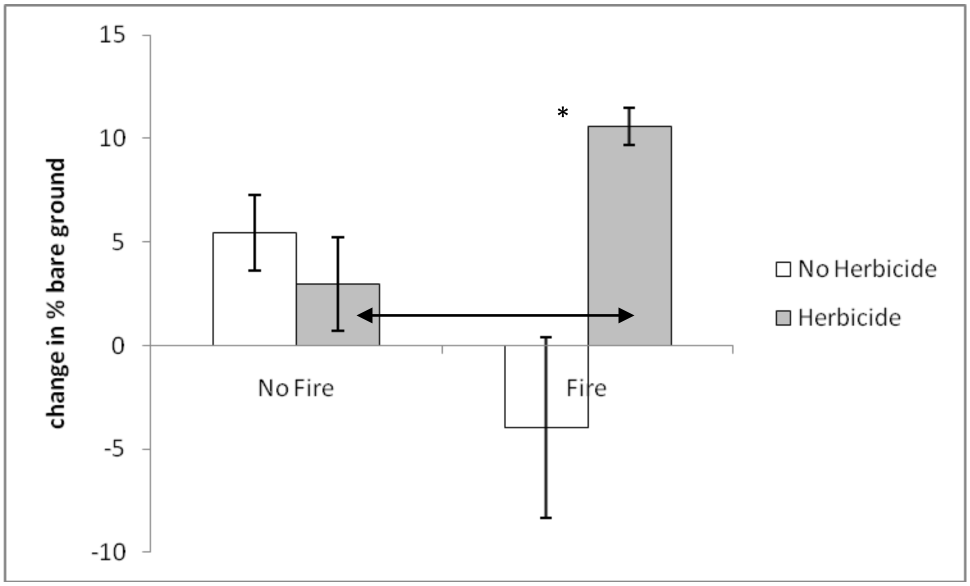

Figure 8. Mean (± 1 SE) difference in percent bare ground cover at the lower site between 2009 (pre-treatment) and 2010 (post-treatment). Arrows indicate significant differences between the two treatment means located at the end points. The asterisk indicates a significant difference between no herbicide and herbicide treatments within a fire treatment.

B. tectorum

The three manipulation treatments reduced *B. tectorum* cover between 2009 and 2010 at the upper site (Table 2); however, the only significant reduction (P=0.0587) occurred with the herbicide treatment. Treatment effects were varied and non-significant at the lower site.

Pre-treatment *B. tectorum* densities were consistently low across the upper and lower sites in 2009, and increased at least 5-fold in 2010 (Table 2). Increases in density were significantly lower in both fire and herbicide treatments (P=0.0205 and P=0.0618, respectively) at the upper site, while treatment effects were non-significant at the lower site.

B. tectorum aboveground biomass decreased in all treatments between 2009 and 2010 at both sites (Table 2). The only significant change in biomass at the upper and lower sites occurred with the herbicide treatment (P=0.0279 and P=0.1009, respectively).

Only the herbicide treatment had a significant impact on *B. tectorum* seed production between 2009 and 2010, decreasing seed numbers (P=0.0195) at both the upper and the lower site (P=0.0652). Mean values for the various treatments at the two sites are shown in Table 2.

Litter and soil seed bank samples were collected in the fall of 2009 to determine the effects of the mowing and fire treatments on seed abundance. The manipulation treatments reduced *B. tectorum* seed density in the litter seed bank at both sites (Table 3); however, at the upper site only the fire treatment was significant (P=0.0802). Seed density at the lower site was also significantly reduced (P=0.0340) by the fire treatment, while there was no reduction associated with the mowing treatment. Even though the manipulation treatments reduced *B. tectorum* seed densities in the soil seed bank by up to 80% at the upper site (Table 3), none of the reductions were significant. At the lower site, seed density in the soil seed bank was significantly reduced (P=0.0339) by the fire treatment. Seed numbers were higher in the litter than in the soil at both sites (Table 3).

Seven manipulation treatments, including 2-way and 3-way combinations of mowing, fire and herbicide were evaluated for their impact on *B. tectorum* seed density in litter and soil seed banks in May 2010. Seed density in the litter seed bank was significantly reduced by the mowing treatment (P<0.0001) at the upper site and by the fire treatment (P=0.0062) at the lower site. Seed density in the soil seed bank was significantly reduced (Table 3) at both the upper site and the lower site by the fire treatment (P=0.0573 and P=0.0183, respectively) and the mowing treatment (P=0.0006 and P=0.0626, respectively). Even though non-significant, there was a trend toward greater reduction in seed density in the litter and soil seed banks when fire was combined with other treatments at both sites.

Annual Forbs

Pre-treatment annual forb cover in 2009 was at least 2 times higher at the upper site than the lower site, leading to larger post-treatment reductions in cover in 2010 (Table 4). Forb cover was significantly reduced (P=0.0910) by the herbicide treatment at the upper site, while the reduction in cover was significant (P=0.0750) for the fire than no-fire treatments at the lower site. In terms of individual species responses, *S. tragus* showed the greatest reduction in cover at the upper site (no data shown).

Table 4. Significant (P<0.1) main-effect means (± 1 SE) for the differences between pre-treatment (2009) and post-treatment (2010) annual forb attributes for the upper and lower sites. Significant means in bold.

Attribute	No Fire	Fire	No Herbicide	Herbicide
Upper Site				
Annual forb cover (%)	-0.91 (4.1)	-0.81 (3.5)	**-0.24 (3.2)**	**-1.5 (4.4)**
Annual forb density (plants/m^2)	**1747 (161)**	**1381 (150)**	1902 (185)	1226 (126)
Annual forb biomass (grams/m^2)	**-9 (3)**	**-6 (3)**	-7 (2)	-9 (4)
Lower Site				
Annual forb cover (%)	**-19 (11)**	**-15 (7)**	-16 (5)	-17 (13)
Annual forb density (plants/m^2)	**805 (51)**	**535 (100)**	1031 (112)	309 (40)
Annual forb biomass (grams/m^2)	-82 (7)	-82 (6)	-97 (7)	-67 (6)

Annual forb density increased in all treatments between 2009 and 2010 at both sites (Table 4). At the upper and lower sites, increases in density were significantly lower (P=0.0342 and P=0.0691, respectively) for fire than no-fire treatments. For individual species, fire had the greatest impact on *S. tragus* and *K. scoparia* at the upper site, and *S. altissimum* at both sites (data not shown).

Pre-treatment annual forb biomass in 2009 was at least 3 times higher at the lower site than the upper site, leading to much larger post-treatment reductions in biomass in 2010 (Table 4). The reduction in biomass was significant (P=0.0178) for fire compared to no-fire treatments at the upper site. The change that occurred within the annual forb biomass at the lower site was affected by both the fire and herbicide treatments as the fire*herbicide was shown to be significant (P=0.0265). For all treatments there was a reduction in biomass of annual forbs; however, with the herbicide treatment, that reduction was smaller especially without fire (Fig. 9).

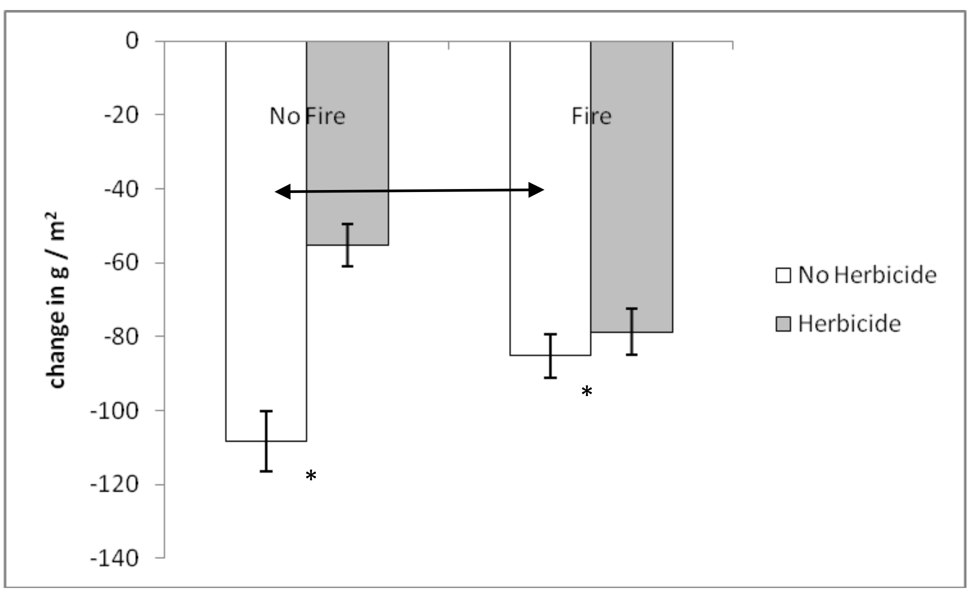

Figure 9. Mean (± 1 SE) difference in annual forb biomass at the lower site between 2009 (pre-treatment) and 2010 (post-treatment). Significance occurred at the fire*herbicide level. Arrows indicate significant differences between the two treatment means located at the end points. Asterisks indicate significant differences between no herbicide and herbicide treatments.

Fire had a major impact on the annual forb seed bank at both sites in fall 2009 (Table 5). It significantly reduced seed density in the litter seed bank (P=0.0144 and P=0.0262, respectively) at the upper and lower sites, and a similar reduction was observed for the soil seed bank (P=0.0224 and P=0.0915, respectively) at the upper and lower sites. Fire had the greatest impact on *S. tragus* and *S. altissimum* seeds in the litter seed bank at the upper site (data not shown).

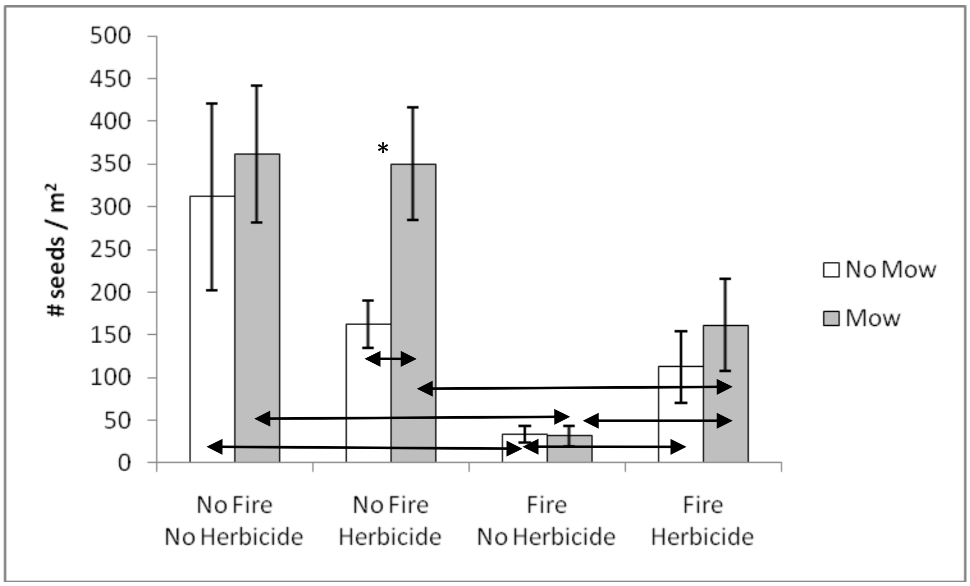

Figure 10. Mean (±1 SE) annual forb seed density in the litter seed bank sampled May 2010 at the upper site. Asterisks indicate significant differences between no mow and mow within a fire* herbicide combination. Arrows indicate significant differences between the two treatment means located at the end points.

The effects of fire and other manipulation treatments on annual forb seed banks were evident at both sites in spring 2010 (Table 5). The fire*herbicide interaction was significant (P=0.0956) for the litter seed bank at the upper site, where the lowest seed densities were observed in the fire-no herbicide treatment combination, followed by the fire-herbicide treatment combination (Fig. 10). The highest seed numbers in the litter seed bank at the upper site were observed in the no fire-no herbicide treatment

Table 5. Significant (P<0.1) main-effect means (± 1 SE) for annual forb and perennial grass seed bank (plants / m^2) in 2009 and 2010 for the upper and lower sites. Significant means in bold, AF=annual forb, PG=perennial grass.

Attribute	No Fire	Fire	No Mow	Mow	No Herbicide	Herbicide
Upper Site						
AF Litter seed bank '09	**10407 (1593)**	**1316 (572)**	5879 (1109)	5844 (1056)	-	-
AF Soil seed bank '09	**504 (114)**	**78 (29)**	322 (83)	260 (60)	-	-
AF Litter seed bank '10	**296 (71)**	**85 (30)**	226 (53)	155 (47)	185 (53)	196 (47)
AF Soil seed bank '10	**52 (16)**	**19 (9)**	32 (10)	38 (14)	37 (13)	33 (12)
PG Litter seed bank '09	25 (15)	10 (5)	10 (7)	24 (13)	-	-
PG Soil seed bank '09	1 (1)	0 (0)	0 (0)	1 (1)	-	-
PG Litter seed bank '10	7 (5)	5 (4)	7 (4)	6 (4)	3 (3)	9 (6)
PG Soil seed bank '10	2 (2)	0.6 (0.7)	0.5 (0.5)	2 (1)	0.6 (0.5)	2 (1)
Lower Site						
AF Litter seed bank '09	**6276 (1410)**	**2449 (397)**	4348 (641)	4377 (1166)	-	-
AF Soil seed bank '09	**555 (115)**	**93 (25)**	411 (96)	237 (44)	-	-
AF Litter seed bank '10	**1194 (309)**	**105 (35)**	866 (263)	433 (81)	782 (516)	250 (94)
AF Soil seed bank '10	82 (25)	38 (13)	**77 (24)**	**42.8 (15)**	78 (25)	41 (14)
PG Litter seed bank '09	307 (94)	88 (56)	236 (116)	159 (35)	-	-
PG Soil seed bank '09	44 (27)	2 (2)	23 (9)	23 (19)	-	-
PG Litter seed bank '10	106 (63)	4(3)	**90 (54)**	**20 (13)**	77 (43)	33 (23)
PG Soil seed bank '10	6 (5)	0.7 (0.7)	2 (2)	4 (3)	3 (2)	3 (3)

combination. Mowing generally increased seed numbers in the various fire-herbicide treatment combinations (Fig. 10). Seed density in the litter seed bank was significantly reduced in the fire treatment (P=0.0266) at the lower site. Fire also significantly reduced seed numbers (P=0.0816) in the soil seed bank at the upper site, while mowing significantly reduced seed numbers (P=0.0780) in the soil seed bank at the lower site. Reductions in seed numbers in the litter seed bank were greatest for *S. tragus* at the upper site and for *S. altissimum* at the lower site (data not shown).

Perennial Grass

The perennial grass functional group was the smallest vegetation component at both sites. Due to low numbers and random distributions across the sites, all but one treatment response for perennial grasses were non-significant. Between 2009 and 2010, there were minimal decreases in cover with manipulation treatments at the upper site and minimal increases in cover at the lower site. Changes in plant density between 2009 and 2010 varied with manipulation treatments at the upper site and decreased with all treatments at the lower site. Aboveground biomass decreased between 2009 and 2010 in all treatments at the upper site, and in all treatments except the fire*herbicide treatement at the lower site. At the upper site in fall 2009, seed numbers were very low in the litter seed bank for all treatments, and there were no seeds in the soil seed bank for three of the four treatments. Perrenial grass seed numbers were higher in the litter and soil seed banks at the lower site in fall 2009, and reductions in seed density were more evident in the manipulation treatments. Seed numbers were significantly affected in the mowing treatment (P=0.0636) at the lower site in spring 2010; similar trends in seed densities in the litter and soil seed banks were observed at both sites (Table 5).

Seeded Species

The effects of the manipulation treatments on the seeded species were evident at both sites (Table 6). Seeded perennial grasses at the upper site showed a significant response to the fire (P=0.0732) and herbicide*mow (P=0.0453) treatments in May 2010, while seeded grasses at the lower site showed a significant response to the fire treatment (P=0.0282). At the upper site the emergence of the plants was largely influenced by the interaction of fire with the other treatments (Fig. 11). With fire, the density of plants was greater than without fire, showing that the presence of the fire treatment had a positive effect on perennial grass germination and survival; however, no significant effects were observed with fire. Plant establishment was greater in the mow-no herbicide treatment than in the mow-herbicide treatment (Fig. 11). In general, the mowing treatment decreased the density of perennial grasses, while it increased with the fire treatment (Table 6). The continued survival of individuals in July 2010 also responded to treatments, as the fire (P=0.0530), mow (P=0.0831) and herbicide*mow (P=0.0253) treatments were all found to be significant at the upper site. The observed interaction of the treatments showed an effect such that the number of individual plants surviving to the July sampling period in areas without fire decreased below 10 plants / m^2 ; while with fire, the number of plants typically stayed above 15 plants / m^2 (Fig. 12). However, in the July sampling period, the observed interaction effect of the herbicide was more pronounced as seen in areas without fire with a decrease in density; where with herbicide, the density of plants was higher as similar patterns were observed as in the May sampling with a decrease in density with the presence of both mowing and herbicide and a general increase with fire in all combinations (Table 6). Fire and herbicide treatments were significant (P=0.0423 and P=0.0126, respectively) at the lower site in July, as both treatments showed better survival than with other treatments. Both sites showed a reduction in perennial grass density between the May and the July sampling periods, as is expected due to seedling mortality.

Table 6. Significant (P<0.1) main-effect means (± 1 SE) for seeded perennial grass density (plants / m^2) in May and July 2010 for the upper and lower sites. Significant means in bold.

Attribute	No Fire	Fire	No Herbicide	Herbicide	No Mow	Mow
Upper Site						
May 2010	**14 (2)**	**27 (3)**	21 (2)	20 (2)	20 (2)	21 (3)
July 2010	**6 (1)**	**17 (2)**	10 (2)	13 (2)	**11 (1)**	**12 (2)**
Lower Site						
May 2010	**10 (1)**	**20 (2)**	13 (1)	17(2)	15 (1)	15 (2)
July 2010	**5 (0.7)**	**12 (2)**	**4 (0.7)**	**13 (2)**	9 (0.8)	9 (2)

Due to low alfalfa densities across all treatments at both sites, it was difficult to identify any trends in treatment effects. Seedling numbers at the upper site for the May and July 2010 sampling periods averaged less than 0.2 plants / m^2, which led to data that could not be normalized. The lower site showed significance only at the July sampling period with the mow and fire*mow treatments (P=0.0168 and P=0.0352, respectively). However, because of the low number of plants observed, the results are inconclusive and are not shown since normality was not achieved.

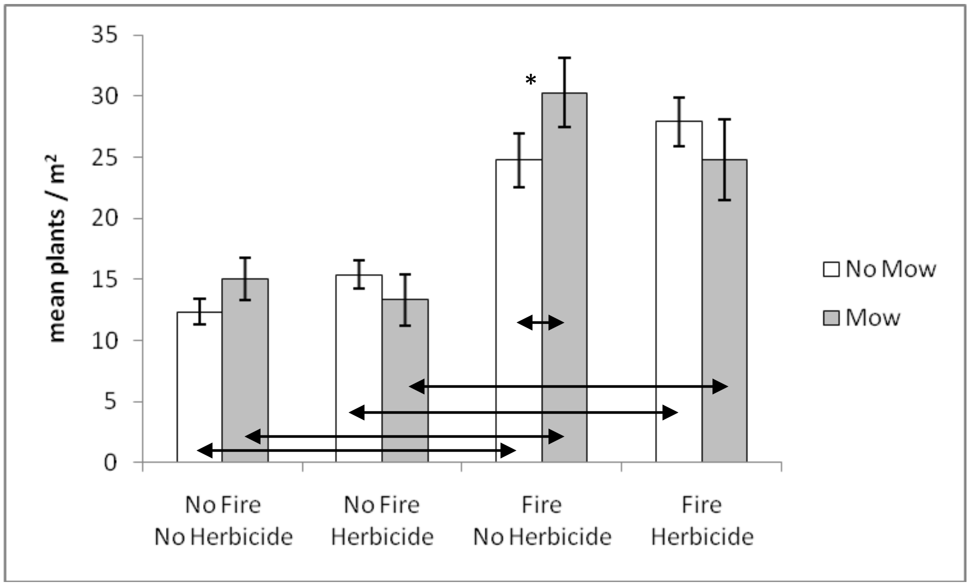

Figure 11. Mean (± 1) seeded perennial grass density at the upper site sampled May 2010. Asterisks indicate significant differences between no mow and mow within a fire * herbicide combination. Arrows indicate significant differences between the two treatment means located at the end points.

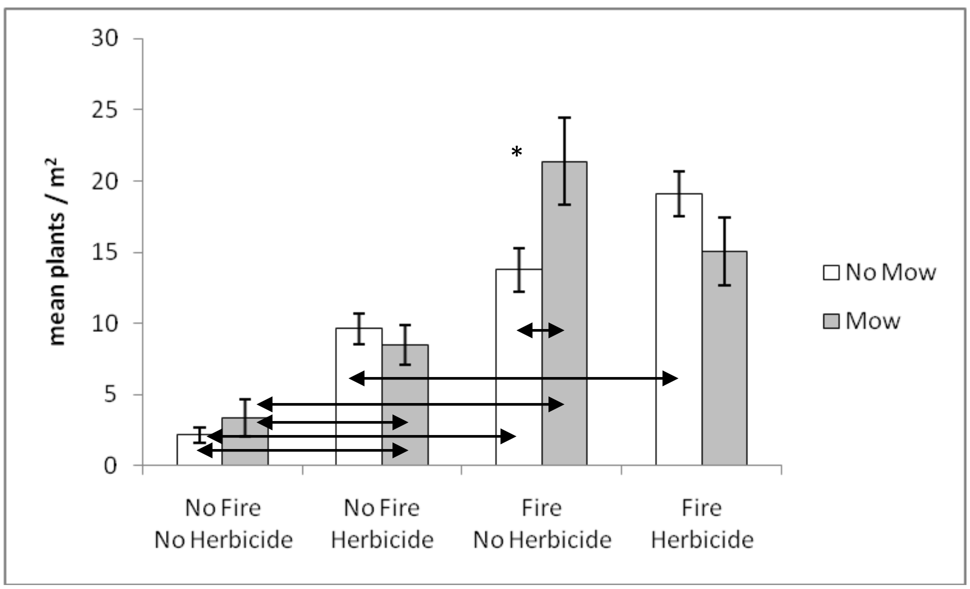

Figure 12. Mean (± 1 SE) seeded perennial grass density at the upper site sampled July 2010. Asterisks indicate significant differences between no mow and mow treatments within a fire * herbicide treatment combination. Arrows indicate significant differences between the two treatment means located at the end points.

DISCUSSION

This study was created to examine the effects of various vegetation manipulation treatments on *B. tectorum*-dominated plant communities as part of a large-scale research demonstration project. The large size enabled us to look at the treatments on a scale that is closer to what actual management would be. This is important because effectiveness or response of a treatment can change from small (one to several m^2) to larger scales (several ha). This study was also developed as part of the EBIPM program, so it can be compared to related studies at similar scales in other areas of the Great Basin. However, because of the large size of the study, there are also some drawbacks, which include the variability in vegetation composition across and between the two sites, and the lack of power available for data analyses with only two replications per experimental unit.

While there were some problems with the size of the study, we were able to accomplish the objective of using the EBIMP framework to evaluate a sequence of manipulation treatments designed to suppress *B. tectorum* and promote the establishment of desirable species in semiarid shrublands in the Great Basin. This framework involves understanding how the three causes of succession alter a system, and how managers can manipulate modifying factors and ecological processes/components to shift vegetation dynamics to a more desirable state.

The first cause of succession is site availability, the presence or absence of safe sites in which seeds can germinate and survive (Harper 1977, Sheley et al. 2010). Site availability is manipulated through controlled disturbances such as fires, disking, and drill seeding. Safe sites for many species include cracks or depressions in the soil, which can collect seed and moisture, or areas where the seed can be covered with a layer of soil (Evans and Young 1972; Evans and Young 1984). In *B. tectorum*-dominated systems, there is typically a layer of litter that collects annual grass seeds and facilitates seedling establishment because of increased moisture retention and temperature moderation at the soil surface; however other species, particularly those with small seeds, cannot work their way through a litter mat like *B. tectorum*, and so the litter becomes a barrier (Evans and Young 1970; Young and Evans 1975; Facelli and Pickett 1991).

In this study, manipulation treatments were implemented to decrease the litter layer that *B. tectorum* prefers, and open sites for desirable resident and seeded species. The mowing treatment (simulating intensive grazing) removed standing live and dead biomass in late-June 2009, and decreased litter input and depth until late-summer 2010 when the next generation of *B. tectorum* plants started to senesce and rebuild the litter layer. Prescribed burning, implemented in early-November 2009, reduced litter depth as a stand-alone treatment and when combined with the mowing treatment, and the mowing*herbicide treatment. However, the fire treatment did not lead to large decreases in litter cover and concomitant large increases in bare ground at both sites. At the time of the burn, the air temperature (14-17 ^{o}C) and relative humidity (23-28%) were barely within the guidelines for conducting a prescribed burn in sagebrush-grasslands in the northern Great Basin (Bunting et al. 1987). In the areas where the fire did reduce litter cover and increase bare ground, particularly at the lower site, there was an associated increase in annual forb density, which is consistent with the findings of Evans and Young (1970). Bare soil allows for increased establishment of annual mustards like *S.altissimum*, because of their ability to conserve water with a mucilaginous seed coat (Young and Evans 1975; Evans and Young 1984).

Seeded perennial grasses were also positively influenced by disturbance associated with drill seeding (Sheley et al. 1996). The seeder creates safe sites by placing seeds at a favorable depth for germination and seedling establishment. The influences of the fire and mowing treatments were also important for the seeded species as the disturbances caused by the treatments removed the biomass from the current year, allowing for a smaller litter layer and reducing the barrier to seedling growth.

Species availability, the second cause of succession, is determined by plant reproductive capacity, seed dispersal mechanisms, and the persistence of seeds in the seed bank (Pickett et al. 2009; Sheley et al. 2010). *B. tectorum* is capable of producing 10-250 seeds/plant on unburned areas and 960-6,000 seeds/plant on burned areas (Young and Evans 1978). Most seeds drop close to the mother plant; however, some seeds can be dispersed long distances by wind, animals and equipment (Young et al. 1987). Dispersed seeds can accumulate in high densities in the soil seed bank, where they can remain viable for at least 3 years (Chepil 1946; Smith et al. 2008; Young and Clements 2009). In contrast, many desirable perennial plants produce less seed and have very limited seed banks in *B. tectorum*-dominated communities (Hassan and West 1986; Humphrey and Schupp 2001). Species availability can be reduced for *B. tectorum* and increased for desirable perennial species with different manipulation methods, including mowing (grazing), fire, herbicide, and drill seeding.

B. tectorum seed production was measured to determine the potential seed rain on the sites, and how it changed with manipulation treatments. This production, pre-seed shatter, resulted in counts ranging from 19,000 to 50,000 seeds/m^2 at the lower site and 20,000 to 27,000 seeds/m^2 at the upper site in 2009, and in counts ranging from 16,000 to 85,000 seeds/m^2 at the lower site, and 20,000 to 71,000 seeds/m^2 at the upper site in 2010. The lower seed numbers at both sites in 2010 were in the herbicide and fire*herbicide treatments, indicating that imazapic applied in fall 2009 reduced the number of potential seed producing plants in the subsequent growing season. *B. tectorum* seed production was measured in a recent study in the northern Great Basin, where Hempy-Mayer and Pyke (2008) found 13,000 to 20,000 seeds/m^2 in a control treatment and only a few hundred to 7,000 seeds/m^2 in a clipping treatment. These findings support the use of grazing animals to reduce *B. tectorum* seed production.

The original experimental design called for intensive cattle grazing to occur from early-May (boot stage) to early-June (purple stage), which has been found to significantly reduce *B. tectorum* seed production in other studies (Mack and Pyke 1984; Pyke 1986; Diamond 2009). This would have involved rotating the cattle through the grazing treatment plots two times in a normal precipitation year. Lack of sufficient cattle numbers and logistical problems hampered implementation of grazing during the first rotation. However, with above-normal growing season precipitation in 2009, the cattle would have been required at the sites until August in order to achieve the utilization goals, and they had to move to federal allotments in mid-June.

Because of the grazing treatment failure, mowing was applied in small areas to simulate grazing defoliation effects. The mowing treatment removed aboveground plant material to a height of 5 cm above the soil surface, while the suction of the lawnmower could have also removed some of the litter layer as well. The above-normal growing season precipitation in 2009 continued late enough in the year so that even after the mowing treatment in late-June, there was enough moisture and time for *B. tectorum* plants to re-grow and set seed. However, seed production was not measured in mowed plots directly, only indirectly through seed bank sampling. Because of the small size of the mowed plots (3 x 3 m), and the lack of trampling associated with grazing, findings from the mowing treatment cannot be readily extended to large-scale settings.

The response of the plants to the disturbances can be related to species availability by the number of seeds produced, but also through the vegetative response of the plants, especially through the bud bank. The disturbance caused by the treatments was intended to reduce the amount of seeds going into the seed bank, thereby reducing the species availability. However, almost all species have the ability to respond vegetatively, such as through rhizomes or buds (Dalgleish and Hartnett 2006). Buds that remain dormant until damage occurs to the plant or environmental conditions allow for activation comprise the bud bank (Lehtila 2000). *B. tectorum* may have the ability to respond to defoliation (the grazing/mowing treatment) through a release of buds, allowing for additional tiller growth. Modeling work by Lehtila (2000) showed that with strong herbivore pressure and late damage and subsequent bud activation, the potential for overcompensation was high in annual plants, leading to even more seed production than may have occurred with undamaged plants. Other studies also found that with regular disturbances caused by fire or grazing, bud activation was stronger and more common than in areas without those disturbances (Benson et al. 2004; Dalgleish and Hartnett 2009). While many studies on bud banks were conducted in tallgrass prairie systems with perennial grass species, it is likely that annual grass systems will respond in a similar manner. Thus, the response of the *B. tectorum* plants to the mowing treatment was likely due to the high precipitation and the availability of the bud banks for additional growth and seed production.

Seed production was not measured for other species; however, the increase in annual forbs at the lower site could be related to post-treatment dispersal events. The common annual forbs at the sites are able to disperse seeds into fire-cleared areas and germinate on the bare soil. *S. altissimum* and *S. tragus* break off close to the soil surface, allowing the plant to tumble with the wind, bouncing seeds loose and increasing species availability (Stallings et al. 1995; Kostivkovsky and Young 2000). The lower site bordered a pasture dominated by *S. altissimum* and *S. tragus*, allowing for increased seed dispersal by these tumbling annual forbs.

Drill seeding perennial species at 7-14 kg/ha PLS at the two sites compensated for the lack of desirable species in sufficient numbers to counteract the existing seed bank dominated by *B. tectorum* and annual forbs. This method of artificial dispersal is a key component in controlling colonization and directing vegetation dynamics (Pickett et al. 2009; Sheley et al. 2010).

Prior to drill seeding desirable perennials, prescribed burning proved to be the most effective treatment for damaging or killing *B. tectorum* seeds within the seed bank. At the lower site, the fire treatment reduced seed density in the litter seed bank by 54% (26,000 seeds/m^2 for control and 12,000 seeds/m^2 for fire) immediately after the burn, November 2009, and by 71% (7,000 seeds/m^2 for control and 400 seeds/m^2 for fire) the following spring, May 2010. Litter and soil seed banks followed similar trends; however, soil numbers were very low compared to those in litter, indicating that the litter seed bank collects most of the seed (Wicks et al. 1971). Humphrey and Schupp (2001) collected seed bank samples (litter and soil combined) on burned and unburned areas immediately after a fire in a *B. tectorum*-dominated community in central Utah, and found 4,800 to 19,000 *B. tectorum* seeds/m^2 in unburned areas, and a 97% reduction in seed density in burned areas. Fire impacts on the *B. tectorum* seed bank at our Park Valley sites might have been greater with more favorable burning conditions. As previously mentioned, the fires did not burn as completely as planned because of unfavorable temperature and relative humidity conditions. There were unburned patches where the entire litter layer was left intact, and in some of the burned patches, undamaged *B. tectorum* seeds were visible underneath the ash, indicating a lower burn temperature than was desired. Fire temperatures needed to be over 150 °C for 5 minutes or 125 °C for 60 minutes to kill the majority of the seeds (Young and Evans 1975; Thill et al. 1984), and that temperature was likely not widely reached.

This study only looked at the seed bank response during the first year after treatment implementation, and future years may not show any effect of the fire treatment due to the rapid recovery of the *B. tectorum* (Humphrey and Schupp 2001). Hassan and West (1986) found a doubling in *B. tectorum* seed production within burned areas compared to unburned areas within one year because of the nutrient resources released by the fire. Other studies indicate that the *B. tectorum* seed bank can recover to pre-fire levels within 2 years because of increased seed production by remaining plants due to increased resource availability (Rasmussen 1994).

Species performance, the third cause of succession, is related to the ability of a plant to respond to surrounding conditions, and is influenced by resource supply, ecophysiology, life history, stress, and interference. The manipulation treatments implemented at the study sites, i.e., mowing, fire, herbicide, and their combinations, were primarily directed at sensitive stages of *B. tectorum* development, i.e., damaging or killing developing seedlings and stressing plants as they begin to flower. By negatively impacting the ability of *B. tectorum* to grow and reproduce, these treatments facilitated seeded species establishment.

Intensive cattle grazing at the boot stage (just before inflorescence emergence from the culm) has been shown to remove biomass and reduce subsequent growth of *B. tectorum* during the remainder of the growing season (Vallentine and Stevens 1994; Diamond 2009). As previously mentioned, cattle grazing in May 2009 was ineffective at reducing *B. tectorum* biomass and seed production at both sites due to cool temperatures and above-normal precipitation. And, the subsequent mowing treatment in late-June 2009 had limited impact on *B. tectorum* performance because there was still enough moisture for the mown plants to regrow and set seed by early-August. The mowed plants, however, were smaller and had fewer seeds than plants developing earlier in the growing season in control plots.

Imazapic herbicide can negatively impact the performance of medusahead (*Taeniatherum caput-medusae*), *B. tectorum*, and several other annual *Bromus* species, especially after a prescribed fire has been applied to reduce the litter layer and expose soil for better herbicide penetration (Monaco et al. 2005; Kyser et al. 2007; Sheley et al. 2007; Morris et al. 2009; Davies 2010; Davies and Sheley 2011). In our study, the effects of imazapic were most evident at the upper site, where the herbicide significantly reduced *B. tectorum* cover, density and biomass. There were greater reductions in these vegetation attributes when imazapic application followed prescribed burning. Most of the *B.*

tectorum plants that did emerge and establish in these treatments appeared to be healthy; however, some plants appeared to have reduced vigor, i.e., stunting and/or signs of injury. Greater suppression of *B. tectorum* may have been achieved with a higher application rate of imazapic (potentially ranging from 105 up to 210 g ai/ha) alone, or in combination with burning, as has been shown in other studies (Monaco et al. 2005; Kyser et al. 2007; Sheley et al. 2007; Morris et al. 2009). When imazapic is applied at rates higher than used in our study (71.6 g ai/ha), there is the potential to injure seeded species (Shinn and Thill 2004; Morris et al. 2009). Because we did not observe any injury to desirable resident species and seeded species at our sites, it would probably be more appropriate to increase the effectiveness of *B. tectorum* suppression by maintaining the same imazapic application rate and burning under more favorable conditions earlier in the fall to expose more bare ground for herbicide contact.

The suppression of *B. tectorum* with these manipulation treatments creates a temporary window of opportunity for the reestablishment of desirable perennial species (Monaco et al. 2005; Davies and Sheley 2011). Density estimates for the seeded perennial grasses in the herbicide, fire, and fire * herbicide treatments during the first year of establishment (2010) met the criteria (at least 10 plants/m^2) for a successful seeding in a semiarid rangeland environment (Vallentine 1989). Their performance in the future will depend on environmental conditions and their ability to compete for resources with new populations of *B. tectorum* and annual forbs. Davies and Sheley (2011) observed that resident native species were promoted by fire with imazapic treatments in *T. caput-medusae* infestations in southeastern Oregon, and suggested that restoration of plant communities invaded by exotic annual grasses may be more successful if efforts focus on areas with some residual native perennial vegetation. We had similar species and amounts (based on cover) of residual perennial grasses in our study sites and did not observe a significant increase in their performance during the first year after implementing similar manipulation treatments.

Other species that had the opportunity to respond positively to the temporary resource availability window were the annual forbs. However, at both sites there was generally a decrease in both cover and biomass of annual forbs, which is contrary to findings from other studies in the northern Great Basin (Evans and Young 1970; Evans et al. 1974). Additional monitoring will show if these trends continue past the first year after treatment.

The final objective of this project was to contribute to the area-wide EBIPM. The mission and goals of the EBIPM group are to increase knowledge about how to best reduce invasive annual grasses utilizing ecologically-based principles and how they relate to individual landscapes. Their desire is to bring together scientists, managers, and landowners in order to best establish and maintain plant communities in order to provide services that are desired. This project examined manipulation treatments influencing the causes of succession, and the findings can be compared to those at similar sites in Idaho, Oregon, and California. Additional work will continue beyond this study and add to the existing knowledge base. The demonstration sites will serve as locations for field tours for managers, land owners and others, and will enable a variety of individuals to observe the impacts of these treatments in *B. tectorum*-dominated communities.

MANAGEMENT IMPLICATIONS

The findings of the study suggest that the use of more than one treatment provides the best results for reducing *B. tectorum* presence in this semiarid system. While very few treatment interactions were significant, the characteristics of the annual system that need to be altered were primarily affected by two different treatments, fire and herbicide. This suggests that for impacts on plant performance, herbicide is most effective, while for site availability (litter reduction) and species availability (seed bank), fire is most effective. The suppression of *B. tectorum* by these manipulation treatments provides a temporary window of opportunity for establishment of desirable species. Preliminary results suggest that revegetation with perennial grass species during this time period increases the potential for establishment while competition from *B. tectorum* and annual forbs is reduced. Follow-up treatments such as imazapic application may be necessary if *B. tectorum* recovery occurs before seeded species get well established. Due to logistical constraints, the prescribed burns were conducted later in the fall (November) than planned, thus reducing the impact of fire and delaying the implementation of the herbicide treatment. We recommend that prescribed burns be conducted earlier in the fall (September) when environmental conditions support a more complete burn, and subsequently allow imazapic application before *B. tectorum* germination.

LITERATURE CITED

Benson, E. J., D.C. Hartnett, and K.H. Mann. 2004. Belowground bud banks and meristem limitation in tallgrass prairie plant populations. American Journal of Botany 91(3): 416-421.

Blumenthal, D.M., U. Norton, J.D. Derner, and J.D. Reeder. 2006. Long-term effects of tebuthiuron on *Bromus tectorum*. Western North American Naturalist 66: 420-425.

Briske, D.D, B.T. Bestelmeyer, T.K. Stringham, and P.L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology & Management 61: 359-367.

Briske, D.D., S.D. Fuhlendorf, and F.E. Smeins. 2005. State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. Rangeland Ecology & Management 58: 1-10.

Briske, D.D., S.D. Fuhlendorf, and F.E. Smeins. 2006. A united framework for assessment and application of ecological thresholds. Rangeland Ecology & Management 59: 225-236.

Bunting, S.C., B.M. Kilgore, and C.L. Bushey. 1987. Guidelines for prescription burning sagebrush-grass rangelands in the northern Great Basin. Ogden, UT, USA: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, General Technical Report INT-231. 33 p.

Call, C.A, J.M. Diamond, and N. Devoe. 2008. Effects of grazing and fire on seed dynamics and species composition of cheatgrass dominated rangeland. Joint meeting of the Society for Range Management and the America Forage and Grassland Council. Poster presentation. Available at: http://srm.confex.com/srm/2008/techprogram/MEETING.HTM. Accessed 01/09/2009.

Carpenter, A.T. and T.A. Murray. 1998. Element Stewardship Abstract for *Bromus tectorum* L. The Nature Conservancy, Arlington, VA. Available at: http://tncweeds.ucdavis.edu/esadocs/documnts/bromtec.pdf. Accessed 11/05/2008. (http://tncweeds.ucdavis.edu.)

Chambers, J.C. 2008. Fire and the Great Basin. In Collaborative management and research in the Great Basin—examining the issues and developing a framework for action. USDA Forest Service, General Technical Report. RMRS-GTR-204. p. 33-37.

Chepil, W.S. 1946. Germination of weed seeds. I. Longevity, periodicity of germination, and vitality of seeds in cultivated soil. Scientific Agriculture 26:307-346.

Cummings, D.C., S.D. Fuhlendorf, and D.M. Engle. 2007. Is altering grazing selectivity of invasive forage species with patch burning more effective than herbicide treatments? Rangeland Ecology & Management 60: 253-260.

Dalgleish, H.J., and D.C. Hartnett. 2006. Below-ground bud banks increase along a precipitation gradient of the North American Great Plains: a test of the meristem limitation hypothesis. New Phytologist 171: 81-89.

Dalgleish, H.J., and D.C. Hartnett. 2009. The effects of fire frequency and grazing on tallgrass prairie productivity and plant composition are mediated through bud bank demography. Plant Ecology 201: 411-420.

Davies, K.W. 2010. Revegetation of medusahead-invaded sagebrush steppe. Rangeland Ecology & Management 63: 564-571.

Davies, K.W., A.M. Nafus, and R.L. Sheley. 2010. Non-native competitive perennial grass impedes the spread of an invasive annual grass. Biological Invasions 12: 3187-3194.

Davies, K.W., and R.L. Sheley. 2011. Promoting native vegetation and diversity in exotic annual grass infestations. Restoration Ecology 19(2): 159-165.

Davison, J. 1996. Livestock grazing in wildland fuel management programs. Rangelands 18: 242-245.

Diamond, J. M. 2009 Effects of targeted grazing and prescribed burning on fire behavior and community dynamics of a cheatgrass (*Bromus tectorum*) dominated landscape. All Graduate Theses and Dissertations. Paper 257. Available at: http://digitalcommons.usu.edu/etd/257. Accessed 1/20/2011.

DiTomaso, J.M., M.L. Brooks, E.B. Allen, R. Minnich, P.M. Rice, and G.B. Kyser. 2006. Control of invasive weeds with prescribed burning. Weed Technology 20: 535-548.

Dooley, S. R., and J. Beckstead. 2010. Characterizing the interaction between a fungal seed pathogen and a deleterious rhizobacterium for biological control of cheatgrass. Biological Control 53: 197-203.

EBIPM. Ecologically-Based Invasive Plant Management. Available at: http://www.ebipm.org/. Accessed 4/18/2011.

Eckert, R. E., Jr., J.E. Asher, M.D. Christensen, and R.A. Evans. 1974. Evaluation of the atrazine-fallow technique for weed control and seedling establishment. Journal of Range Management 27: 288–292.

Epanchin-Niell, R., J. Englin, and D. Nalle. 2009. Investing in rangeland restoration in the arid west, USA: countering the effects of an invasive weed on the long-term fire cycle. Journal of Environmental Management 91: 370-379.

Evans, R. A., and J.A. Young. 1970. Plant litter and establishment of alien annual weed species in rangeland communities. Weed Science 18: 697–703.

Evans, R.A., and J.A. Young. 1972. Microsite requirements for establishment of annual rangeland weeds. Weed Science 20: 350-356.

Evans, R. A., and J.A. Young. 1977. Weed control revegetation systems for big sagebrush-downy brome rangelands. Journal of Range Management 30: 331–336.

Evans, R. A., and J.A. Young. 1984. Microsite requirements for downy brome (*Bromus tectorum*) infestation and control on sagebrush rangelands. Weed Science 32 (Supp. 1): 13–17.

Evans, R. A., R.E. Eckert, Jr., and B.L. Kay. 1967. Wheatgrass establishment with paraquat and tillage on downy brome ranges. Weeds 15: 50–55.

Evans, R. A., H.R. Holbo, R.E. Eckert, Jr., and J.A. Young. 1970. Functional environment of downy brome communities in relation to weed control and revegetation. Weed Science 18: 154–162.

Evans, R.A., J.A. Young, and B.L. Kay. 1974. Germination of winter annual species from a rangeland community treated with paraquat. Weed Science 22: 185-187.

Facelli, J.M., and S.T.A. Pickett. 1991. Plant litter its dynamics and effects on plant community structure. The Botanical Review 57: 1-32.

Harper, J.L. 1977. The population biology of plants. Academic Press. London, UK: 892p.

Harris, G.A. 1967. Some competitive relationships between *Agropyron spicatum* and *Bromus tectorum*. Ecological Monographs 37: 89-111.

Harris, G.A. and A.M. Wilson. 1970. Competition for moisture among seedlings of annual and perennial grasses as influenced by root elongation at low temperature. Ecology 51: 530-534.

Hassan, M.A., and N.E. West. 1986. Dynamics of soil seed pools in burned and unburned sagebrush semi-deserts. Ecology 67: 269-272.

Hempy-Meyer, K., and D.A. Pyke. 2008. Defoliation effects on *Bromus tectorum* seed production: implications for grazing. Rangeland Ecology & Management 61:116-123.

Hild, A.L., M.G. Karl, M.R. Haferkamp, and R.K. Heitschmidt. 2001. Drought and grazing III: root dynamics and germinable seed bank. Journal of Range Management 54: 292-298.

Hulbert, L.C. 1955. Ecological studies *Bromus tectorum* and other annual bromegrasses. Ecological Monographs 25: 181-213.

Hull, A.C., and J.F. Pechanec. 1947. Cheatgrass—a challenge to range research. Journal of Forestry 45: 555-564.

Humphrey, L.D., and E.W. Schupp. 2001. Seed banks of *Bromus tectorum*-dominated communities in the Great Basin. Western North American Naturalist 61: 85-92.

Humphrey, L.D., and E.W. Schupp. 2004. Competition as a barrier to establishment of a native perennial grass (*Elymus elymoides*) in alien annual grass (*Bromus tectorum*) communities. Journal of Arid Environments 58: 405-422.

James, J.J., and T. Svejcar. 2010. Limitations to postfire seedling establishment: the role of seeding technology, water availability, and invasive plant abundance. Rangeland Ecology Management 63: 491-495.

Kay, B. L. 1963. Effects of Dalapon on a medusahead community. Weeds 3: 207–209.

Kay, B. L., and C.M. McKell. 1963. Preemergence herbicides as an aid in seeding annual rangelands. Weeds 11: 260–264.

Kay, B. L., and R.E. Owen. 1970. Paraquat for range seeding in Cismontane California. Weed Science 18(2): 238–244.

Knapp, P.A. 1996. Cheatgrass (*Bromus tectorum* L.) dominance in the Great Basin Desert: history, persistence and influences to human activities. Global Environmental Change 6: 37-52.

Kostivkovsky, V., and J.A. Young. 2000. Invasive exotic rangeland weeds: a glimpse at some of their native habitats. Rangelands 22(6): 3-6.

Kruger-Mangold, J.M., R.L. Sheley, and T.J. Svejcar. 2006. Toward ecologically-based invasive plant management on rangeland. Weed Science 54: 597-605.

Kyser, G.B., J.M. DiTomaso, M.P. Doran, S.B. Orloff, R.G. Wilson, D.L. Lancaster, D.F. Lile, and M.L. Porath. 2007. Control of medusahead (*Taeniatherum caput-medusae*) and other annual grasses with imazapic. Weed Technology 21: 66-75.

Lehtila, K. 2000. Modelling compensatory regrowth with bud dormancy and gradual activation of buds. Evolutionary Ecology 14: 315-330.

Leopold, A. 1949. A Sand County Almanac, and sketches here and there. New York, NY, USA: Oxford University Press. 320 p.

Mack, R.N., and D.A. Pyke. 1984. The demography of *Bromus tectorum*: the role of microclimate, grazing and disease. Journal of Ecology 72: 731-748.

Marsh, N. 1990. "Prescription grazing", an overdue concept. Rangelands 12(2): 101-102.

Melgoza G., R.S. Nowak, and R.J. Tausch. 1990. Soil water exploitation after fire: competition between *Bromus tectorum* (cheatgrass) and two native species. Oecologia 83: 7-13.

Meyer, S.E., D.L. Nelson, S. Clement, and J. Beckstead. 2008. Cheatgrass (*Bromus tectorum*) biocontrol using indigenous fungal pathogens. In: Proceedings-Shrublands under fire: disturbance and recovery in a changing world. Cedar City, UT. Proceedings. RMRS-P-52. S.G. Kitchen, R.L. Pendleton, T.A. Monaco, and J. Vernon, compilers. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, p. 61-67.

Monaco, T.A., T.M. Osmond, and S.A. Dewey. 2005. Medusahead control with fall- and spring-applied herbicides on Northern Utah foothills. Weed Technology 19(3): 653-658.

Morris, C., T.A. Monaco, and C.W. Rigby. 2009. Variable impacts of imazapic rate on downy brome (*Bromus tectorum*) and seeded species in two rangeland communities. Invasive Plant Science and Management 2: 110-119.

Morrow, L.A., and P.W. Stahlman. 1984. The history and distribution of downy brome (*Bromus tectorum*) in North America. Weed Science 32 (Supplement 1): 2-6.

Mosley, J.C. and L. Roselle. 2006. Targeted livestock grazing to suppress invasive annual grasses. In K. Launchbaugh (ED.). Targeted grazing: a natural approach to vegetation management and landscape enhancement. American Sheep Industry Association, Denver, CO. 68-77 p.

NRCS Blue Creek Scan Site. Available at: http://www.wcc.nrcs.usda.gov/nwcc/site?sitenum=2135&state=ut. Accessed April 2010.

Olsen, B.E., and J.H. Richards. 1989. Crested wheatgrass growth and replacement following fertilization, thinning, and neighbor plant removal. Journal of Range Management 42: 93-97.

Olson, R.A., and T.D. Whitson. 2002. Restoring structure in late-successsional sagebrush communities by thinning with tebuthiuron. Restoration Ecology 10: 146-155.

Peeper, T.F. 1984. Chemical and biological control of downy brome (*Bromus tectorum*) in wheat and alfalfa in North America. Weed Science 32 (Supplement 1): 18-25.

Pellant, M., P.L. Shaver, D.A. Pyke, and J.E. Herrick. 2005. Interpreting indicators of rangland health. Version 4. Denver, CO, USA: US Department of the Interior, Bureau of Land Management, Interagency technical reference TR-1734-6. 122 p.

Pickett, S.T.A, S.L. Collins, and J.J. Armesto. 1987. Models, mechanisms, and pathways of succession. Botanical Review 53:335-371.

Pickett, S.T.A., M.L. Cadenasso, and S.J. Meiners. 2009. Ever since Clements: from succession to vegetation dynamics and understanding intervention. Applied Vegetation Science 12:9-21.

Platt, K., and E.R. Jackman. 1946. The cheatgrass problem in Oregon. Oregon State College Extension Bulletin 668: 1-48.

Prevey, J.S., M.J. Germino, N.J. Huntly, and R.S. Inouye. 2010. Exotic plants increase and native plants decrease with loss of foundation species in sagebrush steppe. Plant Ecology 207: 39-51.

Pyke, D.A. 1986. Demographic responses of Bromus tectorum and seedlings of *Agropyron spicatum* to grazing by small mammals: occurrence and severity. Journal of Ecology 74:739-754.

Quadrennial Fire and Fuel Review Report. 2005. Available at: http://www.wildfirelessons.net/documents/QFFR_Final_Report_071905.pdf. Accessed 2/15/2011.

Rasmussen, G. A. 1994. Prescribed burning considerations in sagebrush annual grassland communities. *In*: Monsen, S. B., and S. G. Kitchen, eds. Proceedings: ecology and management of annual rangelands. Ogden, UT: U.S. Department of Agriculture, Forest Service, General Technical Report GTR-INT-313. 69-70.

Reever-Morghan, K.J., R.L. Sheley, and T.J. Svejcar. 2006. Successful adaptive management – the integration of research and management. Rangeland Ecology Management 59: 216-219.

Rice, P.M. 2005. Downy brome: *Bromus tectorum* L. In: C.L. Duncan, and J.K. Clark (EDS.). Invasive plants of range and wildlands and their environmental, economic, and societal impacts. Lawrence, KS. Weed Science Society of America. 147-170.

Rinella, M.J., R.A. Masters, and S.E. Bellows. 2010. Growth regulator herbicides prevent invasive annual grass seed production under field conditions. Rangeland Ecology Management 63: 487-490.

Sheley, R.L., M.F. Carpinelli, and K.J.R. Morghan. 2007. Effects of imazapic on target and nontarget vegetation during revegetation. Weed Technology 21: 1071-1081.

Sheley, R.L., J. James, and E. Vasquez. 2010. Applying ecologically based invasive-plant management. Rangeland Ecology & Management 63: 605-613.

Sheley, R.L., J.M. Mangold, and J.L. Anderson. 2006. Potential for successional theory to guide restoration of invasive-plant-dominated rangeland. Ecological Monographs. 76(3): 365-379.

Sheley, R.L., T.J. Svejcar, and B.D. Maxwell. 1996. A theoretical framework for developing successional weed management strategies on rangeland. Weed Technology 10: 766-773.

Shinn, S.L. and D.C. Thill. 2004. Tolerance of several perennial grasses to imazapic. Weed Technology 18: 60-65.

Simmons, M.T., S. Windhager, P. Power, J. Lott, R.K. Lyons, and C. Schwope. 2007. Selective and non-selective control of invasive plants: the short-term effects of growing-season prescribed fire, herbicide, and mowing in two Texas prairies. Restoration Ecology 15: 662-669.

Smith, D.C., S.E. Meyer, and V.J. Anderson. 2008. Factors affecting *Bromus tectorum* seed bank carryover in Western Utah. Rangeland Ecology & Management 61: 430-436.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. 2008. Web Soil Survey. Available at: http://websoilsurvey.nrcs.usda.gov/. Accessed 11/03/2008. Ecological site description available at: http://www.ut.nrcs.usda.gov/technical/technology/range/mlra28a.html. Accessed 11/03/2008 and 6/25/2009.

Stallings, G.P., D.C. Thill, C.A. Mallory-Smith, and L.W. Lass. 1995. Plant movement and seed dispersal of Russian thistle (*Salsola iberica*). Weed Science 43(1): 63-69.

Stewart, G., and A.C. Hull. 1949. Cheatgrass (*Bromus tectorum* L.) – an ecologic intruder in Southern Idaho. Ecology 30: 58-74.

Stringham, T.K., W.C. Krueger, and P.L. Shaver. 2003. State and transition modeling: an ecological process approach. Journal of Range Management 56: 106-113.

Stubbendieck, J., S.L. Hatch, and L.M. Landholt. 2003. North American wildland plants; a field guide. Lincoln, NE. University of Nebraska Press.

Thill, D.C., K.G. Beck, and R.H. Callihan. 1984. The biology of downy brome (*Bromus tectorum*). Weed Science 32 (Supplement 1): 7-12.

Uresk, D.W., J.F. Cline, and W.H. Rickard. 1979. Growth rates of a cheatgrass community and some associated factors. Journal of Range Management 32: 168-170.

USDA-NRCS. 2009. Plants database. Available at: http://plants.usda.gov/. Accessed 7/7/2009.

Vallentine, J.F. 1989. Range development and improvements. Third edition. San Diego, CA, USA: Academic Press. 524 p.

Vallentine, J.F. and A.R. Stevens. 1994. Use of livestock to control cheatgrass-a review. In: S.B. Monsen and S.G. Kitchen (EDS.) Proceedings-Ecology and management of annual rangelands. 18–21 May 1992. Boise, ID. Ogden UT: USDA-Forest Service Publication. General Technical Report INT-GTR- 313. 202-206.

Vogel, K.P., and R.A. Masters. 2001. Frequency grid: a simple tool for measuring grassland establishment. Journal of Range Management 54: 653-655.

Western Regional Climate Center (WRCC). Available online at: http://www.wrcc.dri.edu/summary/climsmut.html. Accessed 11/3/2008.

Whisenant, S.G. 1990. Changing fire frequencies in Idaho's Snake River plains; Ecological and management implications. In: E.D. Macarther, E.M. Romney, S.D. Smith, and P.T. Tueller [COMPS.]. Proceedings: Symposium on cheatgrass invasion, shrub die-off and other aspects of shrub biology and management, Las Vegas, NV. Ogden, UT: USDA Forest Service Publication. General Technical Report INT-276. 4-10.

Whitson, T.D., and D.W. Koch. 1998. Control of downy brome (*Bromus tectorum*) with herbicides and perennial grass competition. Weed Technology 12: 391-396.

Wicks, G.A., O.C. Burnside, and C.R. Fenster. 1971. Influence of soil type and depth of planting on downy brome seed. Weed Science 19: 82-86.

Wright, H. A. 1974. Range burning. Journal of Range Management 27: 5–11.

Young, J.A., and F.L. Allen. 1997. Cheatgrass and range science. Journal of Range Management 50: 530-535.

Young, J.A., and C.D. Clements. 2007. Cheatgrass and grazing rangelands. Rangelands 29(6): 15-20.

Young, J.A., and C.D. Clements. 2009. Cheatgrass: fire and forage on the range. Reno, Nevada, USA: University of Nevada Press. 348 p.

Young, J.A., and R.A. Evans. 1973. Downy brome—intruder in the plant succession of big sagebrush communities in the Great Basin. Journal of Range Management 26: 410–415.

Young, J.A., and R.A. Evans. 1975. Germinability of seed reserves in a big sagebrush community. Weed Science 23: 358-364.

Young, J.A., and R.A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. Journal of Range Management 31: 283-289.

Young, J.A., and D. McKenzie. 1982. Rangeland drill. Rangelands 4(3): 108–113.

Young, J.A., R.A. Evans, and R.E. Eckert, Jr. 1969. Population dynamics of downy brome. Weed Science 17: 20–26.

Young, J.A., R.A. Evans, and R.E. Eckert, Jr. 1981. Environmental quality and use of herbicides on *Artemisia* grasslands of the U.S. Intermountain area. Agriculture and Environment 6: 53–61.

Young, J.A, R.A. Evans, R.E. Eckert, Jr., and B.L. Kay. 1987. Cheatgrass. Rangelands 9: 266-270.

Young, J.A., R.A. Evans, and J. Major. 1972. Alien plants in the Great Basin. Journal of Range Management 25: 194–201.

Young, J.A., R.A. Evans, and R.A Weaver. 1976. Estimating potential downy brome competition after wildfires. Journal of Range Management 29: 322–325.

Zouhar, K., J.K. Smith, S. Sutherland, and M.L. Brooks. 2008. Wildland fire in ecosystems: fire and nonnative invasive plants. General Technical Report RMRS-GTR-42-vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 355 p. Available at: http://www.fs.fed.us/rm/pubs/rmrs_gtr042_6.pdf. Accessed 13 December 2008 from FEIS site.

APPENDICES

Table A1. Transformations performed on the upper site plant and litter attributes to achieve normality. Not all tests are shown in the Results section due to low numbers and no result.

Upper Site												
	Brte	AF	PG	Pose	Elel	Agcr	Saka	Sial	Kosc	Alfalfa	Bare ground	Litter
cover	square root	log	log	log	log	log	log	log	log	-	square root	square root
density	log	square root	log	square root	log	square root	log	log	square root	-	-	-
biomass	log	log	log	-	-	-	-	-	-	-	-	-
seed production	normal	-	-	-	-	-	-	-	-	-	-	-
litter depth 2009	square root	-	-	-	-	-	-	-	-	-	-	-
litter depth 2010	square root	-	-	-	-	-	-	-	-	-	-	-
litter seedbank 2009	square root	log	square root	-	-	-	square root	log	-	-	-	-
litter seedbank 2010	log	square root	square root	square root	-	-	log	log	-	-	-	-
soil seedbank 2009	log	square root	square root	-	-	-	log	normal	-	-	-	-
soil seedbank 2010	log	square root		-	-	-	square root	normal	-	-	-	-
seeded spp. May	-	-	square root	-	-	-	-	-	-	Not gained	-	-
seeded spp. July	-	-	square root	-	-	-	-	-	-	Not gained	-	-

Table A2. Transformations performed on the lower site plant and litter attributes to achieve normality. Not all tests are shown in the Results section due to low numbers and no result.

Lower Site										
	Brte	AF	PG	Pose	Elel	Saka	Sial	Alfalfa	Bare ground	Litter
cover	square root	log	log	log	log	log	log	-	log	square root
density	log	log	log	log	log	log	square root	-	-	-
biomass	log	log	log	-	-	-	-	-	-	-
seed production	log	-	-	-	-	-	-	-	-	-
litter depth 2009	square root	-	-	-	-	-	-	-	-	-
litter depth 2010	log	-	-	-	-	-	-	-	-	-
litter seedbank 2009	log	log	square root	-	-	square root	square root	-	-	-
litter seedbank 2010	log	log	square root	square root	-	log	log	-	-	-
soil seedbank 2009	log	square root	square root	-	-	square root	log	-	-	-
soil seedbank 2010	log	square root	square root	-	-	square root	square root	-	-	-
seeded spp. May	-	-	square root	-	-	-	-	log	-	-
seeded spp. July	-	-	log	-	-	-	-	log	-	-

Table A3. Results (p-values) of the factorial ANOVA evaluating the differences between pre-treatment (2009) and post-treatment (2010) *B. tectorum* attributes for the upper and lower sites. NS=non-significant.

Attribute	Fire	Herbicide	Fire * Herbicide
Upper Site			
Litter cover (%)	0.0683	NS	NS
Bare ground cover (%)	NS	0.0264	NS
B. tectorum cover (%)	NS	0.0587	NS
B. tectorum density (plants/m^2)	0.0205	0.0618	NS
B. tectorum biomass (grams/m^2)	NS	0.0279	NS
B. tectorum seed production (seed/m^2)	NS	0.0195	NS
Lower Site			
Litter cover (%)	NS	NS	NS
Bare ground cover (%)	0.0443	NS	0.0793
B. tectorum cover (%)	NS	NS	NS
B. tectorum density (plants/m^2)	NS	NS	NS
B. tectorum biomass (grams/m^2)	NS	0.1009	NS
B. tectorum seed production (seed/m^2)	NS	0.0652	NS

Table A4. Results (p-values) of the factorial ANOVA evaluating litter depth and *B. tectorum* seed banks in 2009 and 2010 for the upper and lower sites. NS= non-significant, initials indicate treatment combination types: F=fire, H=herbicide, and M=mow.

Attribute	Fire	Herbicide	Mow	F * H	F * M	H * M	F * H * M
Upper Site							
Litter depth ‘09	NS	NS	0.0416	NS	NS	NS	NS
Litter depth ‘10	NS	NS	0.0001	NS	NS	NS	0.0744
Litter seed bank ‘09	0.0802	NS	NS	NS	NS	NS	NS
Soil seed bank ‘09	NS	NS	NS	NS	NS	NS	NS
Litter seed bank ‘10	NS	NS	0.0001	NS	NS	NS	NS
Soil seed bank ‘10	0.0573	NS	0.0006	NS	NS	NS	NS
Lower Site							
Litter depth ‘09	0.0673	NS	NS	NS	NS	NS	NS
Litter depth ‘10	0.0260	NS	NS	NS	NS	NS	NS
Litter seed bank ‘09	0.0340	NS	NS	NS	NS	NS	NS
Soil seed bank ‘09	0.0339	NS	NS	NS	NS	NS	NS
Litter seed bank ‘10	0.0062	NS	NS	NS	NS	NS	NS
Soil seed bank ‘10	0.0183	NS	0.0626	NS	NS	NS	NS

Table A5. Results (p-values) of the factorial ANOVA evaluating the differences between pre-treatment (2009) and post-treatment (2010) annual forb attributes for the upper and lower sites. NS=non-significant.

Attribute	Fire	Herbicide	Fire * Herbicide
Upper Site			
Annual forb cover (%)	NS	0.0910	NS
Annual forb density (plants/m^2)	0.0342	NS	NS
Annual forb biomass (grams/m^2)	0.0178	NS	NS
Lower Site			
Annual forb cover (%)	0.0750	NS	NS
Annual forb density (plants/m^2)	0.0691	NS	NS
Annual forb biomass (grams/m^2)	NS	NS	0.0265

Table A6. Results (p-values) of the factorial ANOVA evaluating annual forb and perennial grass seed banks in 2009 and 2010 for the upper and lower sites. NS=non-significant, AF= annual forbs, PG= perennial grass, initials indicate treatment combination: F=fire, H=herbicide, M=mow.

Attribute	Fire	Herbicide	Mow	F * H	F * M	H * M	F * H * M
Upper Site							
AF Litter seed bank ‘09	0.0144	NS	NS	NS	NS	NS	NS
AF Soil seed bank ‘09	0.0224	NS	NS	NS	NS	NS	NS
AF Litter seed bank ‘10	0.0515	NS	NS	0.0965	NS	NS	NS
AF Soil seed bank ‘10	0.0816	NS	NS	NS	NS	NS	NS
PG Litter seed bank ‘10	NS	NS	NS	NS	NS	NS	NS
PG Soil seed bank ‘10	NS	NS	NS	NS	NS	NS	NS
Lower Site							
AF Litter seed bank ‘09	0.0262	NS	NS	NS	NS	NS	NS
AF Soil seed bank ‘09	0.0915	NS	NS	NS	NS	NS	NS
AF Litter seed bank ‘10	0.0266	NS	NS	NS	NS	NS	NS
AF Soil seed bank ‘10	NS	NS	0.0780	NS	NS	NS	NS
PG Litter seed bank ‘10	NS	NS	0.0636	NS	NS	NS	NS
PG Soil seed bank ‘10	NS	NS	NS	NS	NS	NS	NS

Table A7. Results (p-values) of the factorial ANOVA evaluating seeded perennial grass density (plants / m^2) at two sampling periods post-treatment (2010) the upper and lower sites. NS=non-significant, initials indicate treatment combination types: F=fire, H=herbicide, and M=mow.

Attribute	Fire	Herbicide	Mow	F * H	F * M	H * M	F * H * M
Upper Site							
May 2010	0.0732	NS	NS	NS	NS	0.0453	NS
July 2010	0.0530	NS	0.0831	NS	NS	0.0253	NS
Lower Site							
May 2010	0.0282	NS	NS	NS	NS	NS	NS
July 2010	0.0423	0.0126	NS	NS	NS	NS	NS

Table A8. Statistical tests for the effects of fire and herbicide on the difference in litter cover between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	13.16	*0.0683
Herb	1	2	5.01	0.1546
Fire*Herb	1	2	0.00	0.9915

Table A9. Statistical tests for the effects of fire and herbicide on the difference in litter cover between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	2.44	0.2584
Herb	1	2	4.66	0.1634
Fire*Herb	1	2	0.24	0.6748

Table A10. Statistical tests for the effects of fire and mowing on litter depth, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.24	0.6706
Mow	1	2	22.53	*0.0416
Fire*Mow	1	2	0.09	0.7894

Table A11. Statistical tests for the effects of fire and mowing on litter depth, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	13.38	*0.0673
Mow	1	2	0.02	0.8959
Fire*Mow	1	2	0.10	0.7769

Table A12. Statistical tests for the effects of fire, herbicide and mowing on litter depth, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.46	0.5678
Herbicide	1	2	0.58	0.5256
Fire*Herbicide	1	2	0.65	0.5059
Mow	1	389	17.92	*<.0001
Fire*Mow	1	389	0.03	0.8666
Herbicide*Mow	1	389	0.53	0.4655
Fire*Herbicide*Mow	1	389	3.20	*0.0744

Table 13. Statistical tests for the effects of fire, herbicide and mowing on litter depth, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	36.92	*0.0260
Herbicide	1	2	1.93	0.2993
Fire*Herbicide	1	2	3.68	0.1951
Mow	1	436	0.78	0.3776
Fire*Mow	1	436	1.70	0.1928
Herbicide*Mow	1	436	0.01	0.9126
Fire*Herbicide*Mow	1	436	0.09	0.7586

Table A14. Statistical tests for the effects of fire and herbicide on the difference in bare ground cover between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	6.60	0.1239
Herb	1	2	36.42	*0.0264
Fire*Herb	1	2	1.88	0.3040

Table A15. Statistical tests for the effects of fire and herbicide on the difference in bare ground cover between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	21.08	*0.0443
Herb	1	2	7.36	0.1132
Fire*Herb	1	2	11.13	*0.0793

Table A16. Statistical tests for the effects of fire and herbicide on the difference in *Bromus tectorum* (BRTE) cover between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	3.39	0.2068
Herb	1	2	15.54	*0.0587
Fire*Herb	1	2	0.65	0.5054

Table A17. Statistical tests for the effects of fire and herbicide on the difference in BRTE cover between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	0.13	0.7489
Herb	1	2	4.41	0.1706
Fire*Herb	1	2	0.10	0.7769

Table A18. Statistical tests for the effects of fire and herbicide on the difference in BRTE density between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	47.28	*0.0205
Herb	1	2	14.69	*0.0618
Fire*Herb	1	2	0.02	0.8889

Table A19. Statistical tests for the effects of fire and herbicide on the difference in BRTE density between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	2.63	0.2466
Herb	1	2	0.90	0.4436
Fire*Herb	1	2	0.03	0.8794

Table A20. Statistical tests for the effects of fire and herbicide on the difference in BRTE biomass between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.04	0.8595
Herbicide	1	2	34.40	*0.0279
Fire*Herbicide	1	2	2.39	0.2625

Table A21. Statistical tests for the effects of fire and herbicide on the difference in BRTE biomass between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	2.58	0.2497
Herbicide	1	2	8.44	*0.1009
Fire*Herbicide	1	2	0.12	0.7647

Table A22. Statistical tests for the effects of fire and herbicide on the difference in BRTE seed production between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.47	0.5628
Herb	1	2	49.81	*0.0195
Fire*Herb	1	2	1.28	0.3749

Table A23. Statistical tests for the effects of fire and herbicide on the difference in BRTE seed production between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	0.00	0.9754
Herb	1	2	13.87	*0.0652
Fire*Herb	1	2	1.12	0.4010

Table A24. Statistical tests for the effects of fire and herbicide on BRTE litter seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	11.00	*0.0802
Mow	1	2	2.72	0.2408
Fire*Mow	1	2	0.01	0.9284

Table A25. Statistical tests for the effects of fire and herbicide on BRTE litter seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	27.96	*0.0340
Mow	1	2	0.21	0.6936
Fire*Mow	1	2	1.05	0.4131

Table A26. Statistical tests for the effects of fire and herbicide on BRTE soil seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	1.49	0.3471
Mow	1	2	2.52	0.2531
Fire*Mow	1	2	0.02	0.8954

Table A27. Statistical tests for the effects of fire and herbicide on BRTE soil seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	28.02	*0.0339
Mow	1	2	0.51	0.5480
Fire*Mow	1	2	0.01	0.9437

Table A28. Statistical tests for the effects of fire, herbicide and mowing on BRTE litter seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	7.31	0.1139
Herb	1	2	3.00	0.2252
Fire*Herb	1	2	0.03	0.8866
Mow	1	95	25.39	*<.0001
Fire*Mow	1	95	0.85	0.3588
Herb*Mow	1	95	0.08	0.7804
Fire*Herb*Mow	1	95	0.01	0.9163

Table A29. Statistical tests for the effects of fire, herbicide and mowing on BRTE litter seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	158.99	*0.0062
Herb	1	2	1.64	0.3289
Fire*Herb	1	2	5.03	0.1542
Mow	1	100	1.72	0.1931
Fire*Mow	1	100	0.13	0.7184
Herb*Mow	1	100	1.21	0.2741
Fire*Herb*Mow	1	100	2.05	0.1558

Table A30. Statistical tests for the effects of fire, herbicide and mowing on BRTE soil seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	15.97	*0.0573
Herb	1	2	0.95	0.4327
Fire*Herb	1	2	0.13	0.7548
Mow	1	85	12.63	*0.0006
Fire*Mow	1	85	0.93	0.3383
Herb*Mow	1	85	0.97	0.3282
Fire*Herb*Mow	1	85	0.56	0.4574

Table A31. Statistical tests for the effects of fire, herbicide and mowing on BRTE soil seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	53.26	*0.0183
Herb	1	2	6.35	0.1280
Fire*Herb	1	2	0.03	0.8777
Mow	1	79	3.57	*0.0626
Fire*Mow	1	79	1.01	0.3191
Herb*Mow	1	79	0.00	0.9452
Fire*Herb*Mow	1	79	0.15	0.7005

Table A32. Statistical tests for the effects of fire and herbicide on the difference in annual forb cover between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	5.29	0.1481
Herb	1	2	9.52	*0.0910
Fire*Herb	1	2	7.71	0.1089

Table A33. Statistical tests for the effects of fire and herbicide on the difference in annual forb cover between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	11.86	*0.0750
Herb	1	2	1.46	0.3505
Fire*Herb	1	2	0.80	0.4657

Table A34. Statistical tests for the effects of fire and herbicide on the difference in annual forb density between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	27.76	*0.0342
Herb	1	2	4.48	0.1686
Fire*Herb	1	2	0.00	0.9899

Table A35. Statistical tests for the effects of fire and herbicide on the difference in annual forb density between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	12.98	*0.0691
Herb	1	2	6.13	0.1317
Fire*Herb	1	2	0.98	0.4265

Table A36. Statistical tests for the effects of fire and herbicide on the difference in annual forb biomass between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	54.68	*0.0178
Herbicide	1	2	5.39	0.1459
Fire*Herbicide	1	2	1.38	0.3604

Table A37. Statistical tests for the effects of fire and herbicide on the difference in annual forb biomass between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	4.70	0.1625
Herbicide	1	2	7.00	0.1181
Fire*Herbicide	1	2	36.25	*0.0265

Table A38. Statistical tests for the effects of fire and herbicide on annual forb litter seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	68.03	*0.0144
Mow	1	2	0.87	0.4491
Fire*Mow	1	2	2.04	0.2894

Table A39. Statistical tests for the effects of fire and herbicide on annual forb litter seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	36.69	*0.0262
Mow	1	2	1.29	0.3744
Fire*Mow	1	2	1.53	0.3419

Table A40. Statistical tests for the effects of fire and herbicide on annual forb soil seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	43.08	*0.0224
Mow	1	2	0.01	0.9300
Fire*Mow	1	2	2.19	0.2772

Table A41. Statistical tests for the effects of fire and herbicide on annual forb soil seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	9.45	*0.0915
Mow	1	2	0.11	0.7756
Fire*Mow	1	2	1.06	0.4107

Table A42. Statistical tests for the effects of fire, herbicide and mowing on annual forb litter seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	17.91	*0.0515
Herb	1	2	2.21	0.2757
Fire*Herb	1	2	8.89	*0.0965
Mow	1	98	2.51	0.1160
Fire*Mow	1	98	1.33	0.2511
Herb*Mow	1	98	1.30	0.2577
Fire*Herb*Mow	1	98	0.21	0.6474

Table A43. Statistical tests for the effects of fire, herbicide and mowing on annual forb litter seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	36.12	*0.0266
Herb	1	2	0.00	0.9747
Fire*Herb	1	2	0.00	0.9569
Mow	1	85	0.35	0.5569
Fire*Mow	1	85	0.70	0.4064
Herb*Mow	1	85	0.81	0.3709
Fire*Herb*Mow	1	85	0.03	0.8599

Table A44. Statistical tests for the effects of fire, herbicide and mowing on annual forb soil seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	10.78	*0.0816
Herb	1	2	0.02	0.8929
Fire*Herb	1	2	2.36	0.2645
Mow	1	100	0.05	0.8244
Fire*Mow	1	100	0.41	0.5233
Herb*Mow	1	100	1.29	0.2581
Fire*Herb*Mow	1	100	0.69	0.4093

Table A45. Statistical tests for the effects of fire, herbicide and mowing on annual forb soil seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	3.06	0.2223
Herb	1	2	1.87	0.3045
Fire*Herb	1	2	0.00	0.9738
Mow	1	100	3.17	*0.0780
Fire*Mow	1	100	0.35	0.5570
Herb*Mow	1	100	0.04	0.8359
Fire*Herb*Mow	1	100	0.68	0.4122

Table A46 Statistical tests for the effects of fire and herbicide on the difference in perennial grass cover between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	1.38	0.3613
Herb	1	2	0.30	0.6403
Fire*Herb	1	2	0.83	0.4574

Table A47. Statistical tests for the effects of fire and herbicide on the difference in perennial grass cover between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.37	0.6027
Herb	1	2	3.32	0.2101
Fire*Herb	1	2	0.30	0.6385

Table A48. Statistical tests for the effects of fire and herbicide on the difference in perennial grass density between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	1.04	0.4150
Herb	1	2	0.87	0.4502
Fire*Herb	1	2	0.80	0.4655

Table A49. Statistical tests for the effects of fire and herbicide on the difference in perennial grass density between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	1	0.42	0.6345
Herb	1	1	1.36	0.4511
Fire*Herb	1	1	0.54	0.5958

Table A50. Statistical tests for the effects of fire and herbicide on the difference in perennial grass biomass between 2009 and 2010 at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.00	0.9981
Herbicide	1	2	0.89	0.4450
Fire*Herbicide	1	2	0.00	0.9936

Table A51. Statistical tests for the effects of fire and herbicide on the difference in perennial grass biomass between 2009 and 2010 at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.83	0.4586
Herbicide	1	2	0.08	0.8052
Fire*Herbicide	1	2	1.20	0.3875

Table A52. Statistical tests for the effects of fire and mow on perennial grass litter seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	1.21	0.3864
Mow	1	2	0.39	0.5947
Fire*Mow	1	2	0.24	0.6736

Table A53. Statistical tests for the effects of fire and mow on perennial grass litter seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	4.35	0.1723
Mow	1	2	0.95	0.4325
Fire*Mow	1	2	0.29	0.6457

Table A54. Statistical tests for the effects of fire and mow on perennial grass soil seed bank, Fall 2009 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.98	0.4272
Mow	1	2	0.98	0.4272
Fire*Mow	1	2	0.98	0.4272

Table A55. Statistical tests for the effects of fire and mow on perennial grass soil seed bank, Fall 2009 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	2.09	0.2854
Mow	1	2	0.00	0.9721
Fire*Mow	1	2	0.26	0.6621

Table A56. Statistical tests for the effects of fire, herbicide, and mowing on perennial grass litter seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	0.09	0.7920
Herb	1	2	1.72	0.3204
Fire*Herb	1	2	0.52	0.5466
Mow	1	98	0.02	0.8762
Fire*Mow	1	98	0.36	0.5517
Herb*Mow	1	98	0.07	0.7905
Fire*Herb*Mow	1	98	1.86	0.1761

Table A57. Statistical tests for the effects of fire, herbicide and mowing on perennial grass litter seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	5.02	0.1545
Herb	1	2	1.59	0.3346
Fire*Herb	1	2	1.04	0.4158
Mow	1	100	3.52	*0.0636
Fire*Mow	1	100	1.77	0.1861
Herb*Mow	1	100	0.30	0.5841
Fire*Herb*Mow	1	100	0.54	0.4641

Table A58. Statistical tests for the effects of fire, herbicide and mowing on perennial grass soil seed bank, Spring 2010 sampling at the upper site.

Type III Tests of Fixed Effects				
Effect	Num DF	Den DF	F Value	Pr > F
Fire	1	2	0.95	0.4326
Herb	1	2	1.10	0.4036
Fire*Herb	1	2	0.00	0.9876
Mow	1	100	1.38	0.2429
Fire*Mow	1	100	0.01	0.9154
Herb*Mow	1	100	0.03	0.8553
Fire*Herb*Mow	1	100	0.78	0.3780

Table A59. Statistical tests for the effects of fire, herbicide and mowing on perennial grass soil seed bank, Spring 2010 sampling at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	4.14	0.1788
Herb	1	2	0.01	0.9461
Fire*Herb	1	2	0.53	0.5427
Mow	1	100	0.72	0.3974
Fire*Mow	1	100	0.02	0.9021
Herb*Mow	1	100	0.10	0.7554
Fire*Herb*Mow	1	100	1.08	0.3013

Table A60. Statistical tests for the effects of fire, herbicide and mowing on perennial grass May 2010 seeded species data at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	12.17	*0.0732
Herb	1	2	0.00	0.9901
Fire*Herb	1	2	0.05	0.8493
Mow	1	340	1.03	0.3110
Fire*Mow	1	340	0.03	0.8519
Herb*Mow	1	340	4.04	*0.0453
Fire*Herb*Mow	1	340	0.09	0.7634

Table A61. Statistical tests for the effects of fire, herbicide and mowing on perennial grass May 2010 seeded species data at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	33.94	*0.0282
Herb	1	2	8.31	0.1022
Fire*Herb	1	2	1.56	0.3383
Mow	1	340	1.26	0.2621
Fire*Mow	1	340	0.59	0.4437
Herb*Mow	1	340	0.10	0.7564
Fire*Herb*Mow	1	340	1.02	0.3138

Table A62. Statistical tests for the effects of fire, herbicide and mowing on perennial grass July 2010 seeded species data at the upper site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	17.40	*0.0530
Herb	1	2	3.46	0.2038
Fire*Herb	1	2	2.59	0.2488
Mow	1	340	3.02	*0.0831
Fire*Mow	1	340	0.12	0.7286
Herb*Mow	1	340	5.05	*0.0253
Fire*Herb*Mow	1	340	1.52	0.2191

Table A63. Statistical tests for the effects of fire, herbicide and mowing on perennial grass July 2010 seeded species data at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	22.15	*0.0423
Herb	1	2	77.95	*0.0126
Fire*Herb	1	2	2.70	0.2423
Mow	1	277	2.42	0.1206
Fire*Mow	1	277	1.88	0.1719
Herb*Mow	1	277	2.14	0.1445
Fire*Herb*Mow	1	277	0.28	0.5969

Table A64. Statistical tests for the effects of fire, herbicide and mowing on alfalfa May 2010 seeded species data at the lower site.

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	2	1.93	0.2996
Herb	1	2	2.84	0.2338
Fire*Herb	1	2	0.08	0.8009
Mow	1	82	2.37	0.1276
Fire*Mow	1	82	1.90	0.1714
Herb*Mow	1	82	0.01	0.9364
Fire*Herb*Mow	1	82	0.07	0.7986

Table A65. Statistical tests for the effects of fire, herbicide and mowing on alfalfa July 2010 seeded species data at the lower site. (values not shown were not able to be calculated due to low numbers)

Type III Tests of Fixed Effects				
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Fire	1	1	0.35	0.6611
Herb	1	1	2.33	0.3693
Fire*Herb	0	.	.	.
Mow	1	52	6.11	*0.0168
Fire*Mow	1	52	4.67	*0.0352
Herb*Mow	1	52	0.08	0.7766
Fire*Herb*Mow	0	.	.	.

Table A66. Results of statistical tests at the upper site. Significance at P=0.1. Brte= *Bromus tectorum*, AF=annual forbs, PG =perennial grass, Satr=*Salsola tragus*, Sial=*Sisymbrium altissimum*, Kosc=*Kochia scoparia*.

Upper Site								
	Brte	AF	PG	Satr	Sial	Kosc	Bare ground	Litter
cover	herbicide	herbicide	-	fire	-	-	herbicide	fire
density	herbicide, fire	fire	-	fire	herbicide	herbicide		
biomass	herbicide	fire	-					
seed production	herbicide							
litter depth 2009	mow							
litter depth 2010	mow, F*H*M							
litter seedbank 2009	fire	fire	-	fire	fire			
litter seedbank 2010	mow	fire, F*H		fire	-			
soil seedbank 2009	-	fire	-	-	-			
soil seedbank 2010	fire, mow	fire	-	-	-			
seeded May			fire, H*M					
seeded July			fire, mow, H*M					

Table A67. Results of statistical tests at the lower site. Significance at P=0.1. Brte= *Bromus tectorum*, AF=annual forbs, PG =perennial grass, Satr=*Salsola tragus*, Sial=*Sisymbrium altissimum*.

Lower Site								
	Brte	AF	PG	Satr	Sial	Alfalfa	Bare ground	Litter
cover	-	fire	-	-	-		fire, F*H	-
density	-	fire	-	-	herbicide			
biomass	herbicide	F*H	-					
seed production	herbicide							
litter depth 2009	fire							
litter depth 2010	fire							
litter seedbank 2009	fire	fire	-	mow	-			
litter seedbank 2010	fire, mow	fire	mow	-	fire, H*M			
soil seedbank 2009	fire	fire	-	-	-			
soil seedbank 2010	fire, mow	mow	-	-	mow			
seeded May			fire			-		
seeded July			fire, herbicide			mow, F*M		

Appendix B

	Sampled Species	
	scientific name	common name
Annual Grasses		
	Bromus tecorum	cheatgrass
	Vulpia octoflora	sixweeks fescue
Annual Forbs		
	Chenopodium spp.	goosefoot/lambsquarters
	Cryptantha rugulosa	blue-eyed susan
	Descurainia pinnata	tansy mustard
	Erodium cicutarium	redstem filaree
	Gilia spp.	gilia
	Halogeton glomeratus	halogeton
	Kochia scoparia	annual kochia
	Latuca serrolia	prickly lettuce
	Lepidium perfoliatum	clasping pepperweed
	Ceratocephala testiculatus	bur buttercup
	Salsoli tragus	Russian thistle
	Sisymbrium altissimum	tumble mustard
	Tragopogon dubius	western salsify
Perennial Grasses		
	Achnatherum hymenoides	indian ricegrass
	Agropyron cristatum	crested wheatgrass
	Elymus elymoides	bottlebrush squirreltail
	Poa secunda	Sandberg's bluegrass
Perennial Forbs		
	Calochortus nuttallii	sego lily
	Oenothera caespitosa	evening primrose
	Phlox spp.	phlox
	Spaeralcea spp.	globemallow
	Opuntia polyacantha	pricklypear cactus
Perennial Shrubs		
	Chrysothamnus viscidiflorus	yellow rabbitbrush
	Sarcobatus vermiculatus	greasewood